David J. Erickson, PG, CPG

President/Hydrogeologist
Water & Environmental Technologies, PC
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Education

- Bachelor of Science, Geological Engineering, Montana College of Mineral Science & Technology 1988
- Continuing Education Credits – 1990, 1991

Professional History

- *Water & Environmental Technologies*; Butte, MT, President/Hydrogeologist, August 2000 – present
- *Atlatl, Inc*., Butte; MT, Principal Hydrogeologist/Project Manager, May 1994 – August 2000
- *Special Resource Management, Inc.;* Butte, MT, Geological Engineer/Hydrogeologist, 1990-1994
- *Woodward-Clyde Consultants;* Houston, Texas, Staff Geological Engineer/Hydrogeologist, 1989-1990
- *Petroleum Testing Service;* Houston, Texas, Geological Technician, 1988-1989

Representative Experience

Project Manager and Hydrogeologist responsible for the characterization and remediation of a dissolved solvent plume from a county landfill. Remediation consists of in-situ air sparging and a funnel-and-gate capture and in-situ treatment system. The sites complex fractured bedrock and extremely complex ground water flow characteristics required innovative investigation technology to understand the water and contaminant interaction between the bedrock and the alluvial aquifers and ground water and surface water. Project highlights include:

- The use of geophysical method to characterize the bedrock topography and the connection and interaction between aquifers,
- The use of direct push subsurface investigation methods to characterize site conditions and identify contaminant transport pathways,
- Ground water flow and contaminant transport modeling to describe site conditions and test remedial options,
- The installation of source specific remedial methods to control landfill leachate impacts,
- Long term responsibility for all surface water, ground water, remediation, and reporting requirements for the site, and
- Presentation of site characteristics, model results, and site remediation costs in District Court.

Project Hydrogeologist and Lead Expert for the investigation and characterization of geologic, hydrogeologic, and contaminant migration characteristics of solvent and fuel contamination impacting a residential neighborhood. The goal of the investigation work was to determine the source of contamination and identify the responsible party. Geophysical methods (soil conductivity logging) and depth specific profile sampling was used to identify perchloroethylene migration and degradation in multiple production zones within the alluvial aquifer. This subsurface investigation established a connection between historical lagoon leakage and residential supply wells.

Project Manager and Lead Expert conducting a site investigation to assess the impact of historical mining and milling activities on ground water and stream water quality. Dissolved metals concentrations impacting a small town public water supply system prompted a complaint against the Mining Company. Tailings investigations and in stream tracer testing established a direct connection between stream water contamination and spring contamination.

Project Hydrogeologist/Manager for the investigation and remediation of many UST and Hazardous Waste Sites. Contaminants include fuels, solvents, wood treating compounds, metals, pesticides, herbicides, fungicides, and fertilizers.

Project Manager/Hydrogeologist responsible for the design, installation, and monitoring of various types of remedial technologies or remedial methods including (air stripping, air sparging, vapor extraction, bioventing, bio-cell treatment, biostimulation (ORC), NAPL recovery, in-situ & ex-situ bioremediation, natural attenuation, excavation & off-site disposal).

Project Manager responsible for the investigation and remediation of 29 sites in Montana and North Dakota where pesticides, herbicides, fungicides, fuels and fertilizers were spilled.

Project Manager and Hydrogeologist for extensive study and ground water modeling of contaminant effects from ash disposal ponds on an arid Wyoming drainage. The study involved:

- Prediction of contaminant transport,
- Simulation of remedial options,
- Design, installation, optimization and operation of remediation system,
- Permitting of facility expansion,
- Extensive presentations and negotiations with regulatory agencies, and
- Dispute resolution between the facility and potentially effected parties.

Project Engineer responsible for the design and permitting of a double-lined hazardous and non-hazardous repository with leachate collection and ground water relief system.

Project Engineer and Project Manager responsible for the design of ground water monitoring systems and subsurface geological, hydrogeological, and geotechnical investigation.

Project Hydrogeologist studying ground water fluctuations at a RCRA Part B TSD (Hazardous Waste Disposal Facility) in Oregon. Both hydrogeologic and contaminant transport characteristics were very complex.

Project Hydrologist responsible for sediment transport and stream water quality modeling for mine tailing disposal project in Malasia.

Project Hydrogeologist responsible for re-permitting several industrial landfills for large coal-fired electric generating plants in Wyoming. Projects involved investigation of water quality degradation from fly ash disposal activities and characterization of the potential health risks. A statistical evaluation of the water quality was completed to identify potential impacts.

Project Hydrogeologist for evaluation water chemistry changes resulting from the use of wastewater for irrigation at a research farm in Utah.

Project Hydrogeologist for yearly monitoring data analysis at several industrial plants with ponds or landfills in Wyoming and Utah.

Project Hydrogeologist performing final phase of landfill siting study for new RCRA Subtitle D Municipal Solid Waste Landfill

Project Hydrogeologist/Manager for the investigation and remediation of many UST and Hazardous Waste Sites. Contaminants include fuels, solvents, wood treating compounds, metals, pesticides, herbicides, fungicides, and fertilizers.

Project Manager/Hydrogeologist responsible for the design, installation, and monitoring of various types of remedial technologies or remedial methods including (air stripping, air sparging, vapor extraction, bioventing, bio-cell treatment, biostimulation (ORC), NAPL recovery, in-situ & ex-situ bioremediation, natural attenuation, excavation & off-site disposal).

Project Manager responsible for the investigation and remediation of 29 sites in Montana and North Dakota where pesticides, herbicides, fungicides, fuels and fertilizers were spilled.

Expert Witness/Litigation Support Experience

- *Park County v. Burlington Northern Santa Fe Railway Company, Montana Sixth Judicial District Court, Park County, Cause No. DV 97-75, July, 1999.*

- *C&P Packing v. Burlington Northern Santa Fe Railway Company*, Park County, January 2001.
- *Hepp v. Conoco Inc. et. al.*, ADV-2003-14
- *Town of Sunburst v. Texaco et. al.*, CDV-01-179 (a)
- *Town of Superior v. Asarco Incorporated*, US District Court, Missoula Division
- *Aguiar v. Burlington Northern, United States District Court, Great Falls Division*
- *Schammel et. al. v.CR Kendall Corporation, United States District Court, Great Falls Division.*
- *Van Haur v. CR Kendal Corp United States District Court, Great Falls Division*
- *Weiss et. al. v. HCI Dyce Chemical Company, CV-00-123-BLG-JDS*
- *Sieben Livestock Company v. Harp Line Contractors.*
- *Cool Breeze Inc. v. Flying J Inc., Maxim Technologies Inc.*
- *Cause No. ADV-04-984*
- *Friends of the Little Bitterroot v. Commissioners of Flathead County Cause No.: DV-06-560*
- *Mapleton City Corporation v. The Ensign-Bickford Company*, Case No. 020404933
- *Bergren v. BNSF*: CV-03-120-BLG-RFC
- *Devries v. BNSF:* CV-03-121-BLG-RFC
- *Outlook Enterprises v. BNSF*: CV-03-139-BLG-RFC
- *Hallett Minerals v. BNSF* Cause No. CV-03-161-BLG-RFC
- *Ruggles Excavation v. BNSF* Cause No. CV-03-160-BLG-RFC
- *Burley, Nelson, Meridith v. BNSF*
- *Anderson et. al. v. BNSF Cause No. ADV-2008-101*
- *Kerfoot v. Texaco et. al. Cause No BDV-08-1276*
- *City of Livingston et. al. V. BNSF, Cause No. DV07-141*
- *Graham et, al.v. BNSF, Cause No. CV-12-145-M-DVM*

Professional Development

- Hazardous Waste and Geotech Sampling Seminar
- Monitoring Well Installation Seminar
- Analytical Laboratory Seminar (ENSECO)
- Design & Construction of RCRA Final Covers
- Enhanced Bioremediation (EPA)
- Ground Water Pollution & Hydrogeology, Princeton
- Geostatistical Analysis in Hazardous Waste Site Evaluation
- Ground Water Summit 2008
- Montana Water Law Conference 2007
- Landfill Gas Extraction & Ground Water Corrective Measures (presenter)
- National Ground Water Association Annual Conference – heterogeneity
- Environmental Geochemistry of Metals
- Environmental Isotopes in Ground Water Resource and Environmental Contamination
- Environmental Forensics: Methods & Applications
- 2004 NGWA Water & Environmental Law Conference

Certifications
Professional Geologist, Wyoming PG-3101
Professional Geologist, Utah PG-2250
Certified Professional Geologist, American Institute of Professional Geologists, CPG#9402
OSHA 29 CFR 1910.120 Health & Safety
OSHA 29 CFR Certified Waste Site Supervisor
Certified Monitoring Well Constructor

Affiliations
Association of Ground Water Scientists & Engineers
National Ground Water Association
American Institute of Professional Geologist
American Chemical Society
International Society of Environmental Forensics

Awards
Montana Tech Distinguished Alumni Recognition Award, 2003

Seepage from Cow Palace Waste Lagoons

Flow Boundary Conditions

Steady-state flow
Constant hydraulic head (h) of liquid-wastes level in the lagoon for given calculation
(i.e. no change of head in time)
No flow restriction at the bottom of liner (like a "free fall")

Formula used

q = k(h/d)
Q = k(h/d)A
h = w+d

Conversion factors

1 ft	30.48	cm	304.8 mm
1 ft^3	7.48	gal	
1 day	86,400	s	
1 year	365	days	
1 GPY/sqft	0.11	mm/d	
1 Acre	43,546	sqft	

Assumptions & Entry Parameters

Liquid waste is homogenous in vertical and horizontal planes			
Specific gravity of liquid wastes	1	(-)	
Lagoon No. 4 liner			
Hydraulic conductivity (K) equals	1.00E-06	cm/s	
Thickness (d)	2	ft	
Other lagoons			
Hydraulic conductivity (K) of a liner to be installed		cm/s	[independent variable]
Thickness (d) of liner to be installed		ft	[independent variable]
Depth of liquid waste (w)		ft	[independent variable]

Seepage (q) from lined lagoon per unit area (w/o self sealing)

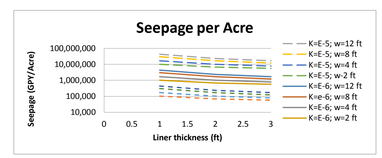
Calculations performed as per:

"Design and Construction Guidelines for Considering Seepage from Agricultural Waste Storage Ponds and Treatment Lagoons", 1993 Soil Conservation Service , U.S. Department of Agriculture

K (cm/s)	w (ft)	d (ft)	Seepage GPY/Acre	Seepage mm/day
1.E-05	12	1	43,811,171	112.3
		2	23,590,631	60.5
		3	16,850,450	43.2
	8	1	30,330,811	77.8
		2	16,850,450	43.2
		3	12,356,997	31.7
	4	1	16,850,450	43.2
		2	10,110,270	25.9
		3	7,863,544	20.2
	2	1	10,110,270	25.9
		2	6,740,180	17.3
		3	5,616,817	14.4

K (cm/s)	w (ft)	d (ft)	Seepage GPY/Acre	Seepage mm/day
1.E-06	12	1	4,381,117	11.2
		2	2,359,063	6.0
		3	1,685,045	4.3
	8	1	3,033,081	7.8
		2	1,685,045	4.3
		3	1,235,700	3.2
	4	1	1,685,045	4.3
		2	1,011,027	2.6
		3	786,354	2.0
	2	1	1,011,027	2.6
		2	674,018	1.7
		3	561,682	1.4

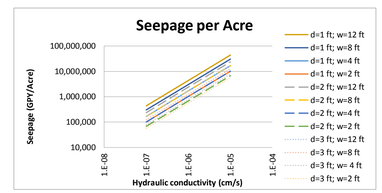
K (cm/s)	w (ft)	d (ft)	Seepage GPY/Acre	Seepage mm/day
1.E-07	12	1	438,112	1.12
		2	235,906	0.60
		3	168,505	0.43
	8	1	303,308	0.78
		2	168,505	0.43
		3	123,570	0.32
	4	1	168,505	0.43
		2	101,103	0.26
		3	78,635	0.20
	2	1	101,103	0.26
		2	67,402	0.17
		3	56,168	0.14



d (ft)	w (ft)	K (cm/s)	Seepage GPY/Acre	Seepage mm/day
1.0	12	1.E-07	438,112	1.1
		1.E-06	4,381,117	11.2
		1.E-05	43,811,171	112.3
	8	1.E-07	303,308	0.8
		1.E-06	3,033,081	7.8
		1.E-05	30,330,811	77.8
	4	1.E-07	168,505	0.4
		1.E-06	1,685,045	4.3
		1.E-05	16,850,450	43.2
	2	1.E-07	101,103	0.3
		1.E-06	1,011,027	2.6
		1.E-05	10,110,270	25.9

d (ft)	w (ft)	K (cm/s)	Seepage GPY/Acre	Seepage mm/day
2.0	12	1.E-07	235,906	0.6
		1.E-06	2,359,063	6.0
		1.E-05	23,590,631	60.5
	8	1.E-07	168,505	0.4
		1.E-06	1,685,045	4.3
		1.E-05	16,850,450	43.2
	4	1.E-07	101,103	0.3
		1.E-06	1,011,027	2.6
		1.E-05	10,110,270	25.9
	2	1.E-07	67,402	0.2
		1.E-06	674,018	1.7
		1.E-05	6,740,180	17.3

d (ft)	w (ft)	K (cm/s)	Seepage GPY/Acre	Seepage mm/day
3.0	12	1.E-07	168,505	0.4
		1.E-06	1,685,045	4.3
		1.E-05	16,850,450	43.2
	8	1.E-07	123,570	0.3
		1.E-06	1,235,700	3.2
		1.E-05	12,356,997	31.7
	4	1.E-07	78,635	0.2
		1.E-06	786,354	2.0
		1.E-05	7,863,544	20.2
	2	1.E-07	56,168	0.1
		1.E-06	561,682	1.4
		1.E-05	5,616,817	14.4

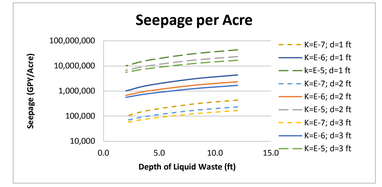


d (ft)	K (cm/s)	w (ft)	Seepage GPY/Acre	Seepage mm/day
1.0	1.E-07	12.0	438,112	1.1
		8.0	303,308	0.8
		4.0	168,505	0.4
		2.0	101,103	0.3
	1.E-06	12.0	4,381,117	11.2
		8.0	3,033,081	7.8
		4.0	1,685,045	4.3
		2.0	1,011,027	2.6
	1.E-05	12.0	43,811,171	112.3
		8.0	30,330,811	77.8
		4.0	16,850,450	43.2
		2.0	10,110,270	25.9

d (ft)	K (cm/s)	w (ft)	Seepage GPY/Acre	Seepage mm/day
2.0	1.E-07	12.0	235,906	0.6
		8.0	168,505	0.4
		4.0	101,103	0.3
		2.0	67,402	0.2
	1.E-06	12.0	2,359,063	6.0
		8.0	1,685,045	4.3
		4.0	1,011,027	2.6
		2.0	674,018	1.7
	1.E-05	12.0	23,590,631	60.5
		8.0	16,850,450	43.2
		4.0	10,110,270	25.9
		2.0	6,740,180	17.3

Brown highlighted cells show seepage from Lagoon 4

d (ft)	K (cm/s)	w (ft)	Seepage GPY/Acre	Seepage mm/day
3.0	1.E-07	12.0	168,505	0.4
		8.0	123,570	0.3
		4.0	78,635	0.2
		2.0	56,168	0.1
	1.E-06	12.0	1,685,045	4.3
		8.0	1,235,700	3.2
		4.0	786,354	2.0
		2.0	561,682	1.4
	1.E-05	12.0	16,850,450	43.2
		8.0	12,356,997	31.7
		4.0	7,863,544	20.2
		2.0	5,616,817	14.4



Seepage (q) from self sealed lagoon per unit area

Flow Boundary Conditions

Same as for the calculations for liners (above)

Calculations performed as per:

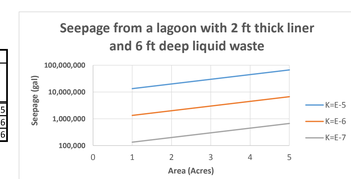
Selected references claiming that Animal Waste Lagoons may self-seal up to K
one order of magnitude lower than the original sediment at the lagoon bottom

Assumptions & Entry Parameters

Hydraulic conductivity (Ks) of a self-sealed layer	cm/s	[independent variable]
Thickness of a self-sealed layer (ds)	ft	[independent variable]
Assumptions are the same as for the calculation for liners (above)		

Seepage from a lagoon with a 2 ft thick liner

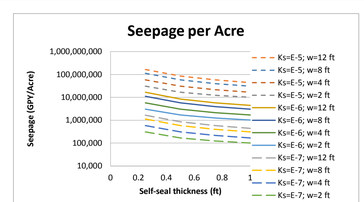
d (ft)	w (ft)	K (cm/s)	Seepage GPY per 1 acre	Seepage GPY per 5 acres	Seepage mm/day
2.0	6	1.E-07	134,804	674,018	0.35
		1.E-06	1,348,036	6,740,180	3.46
		1.E-05	13,480,360	67,401,802	34.56

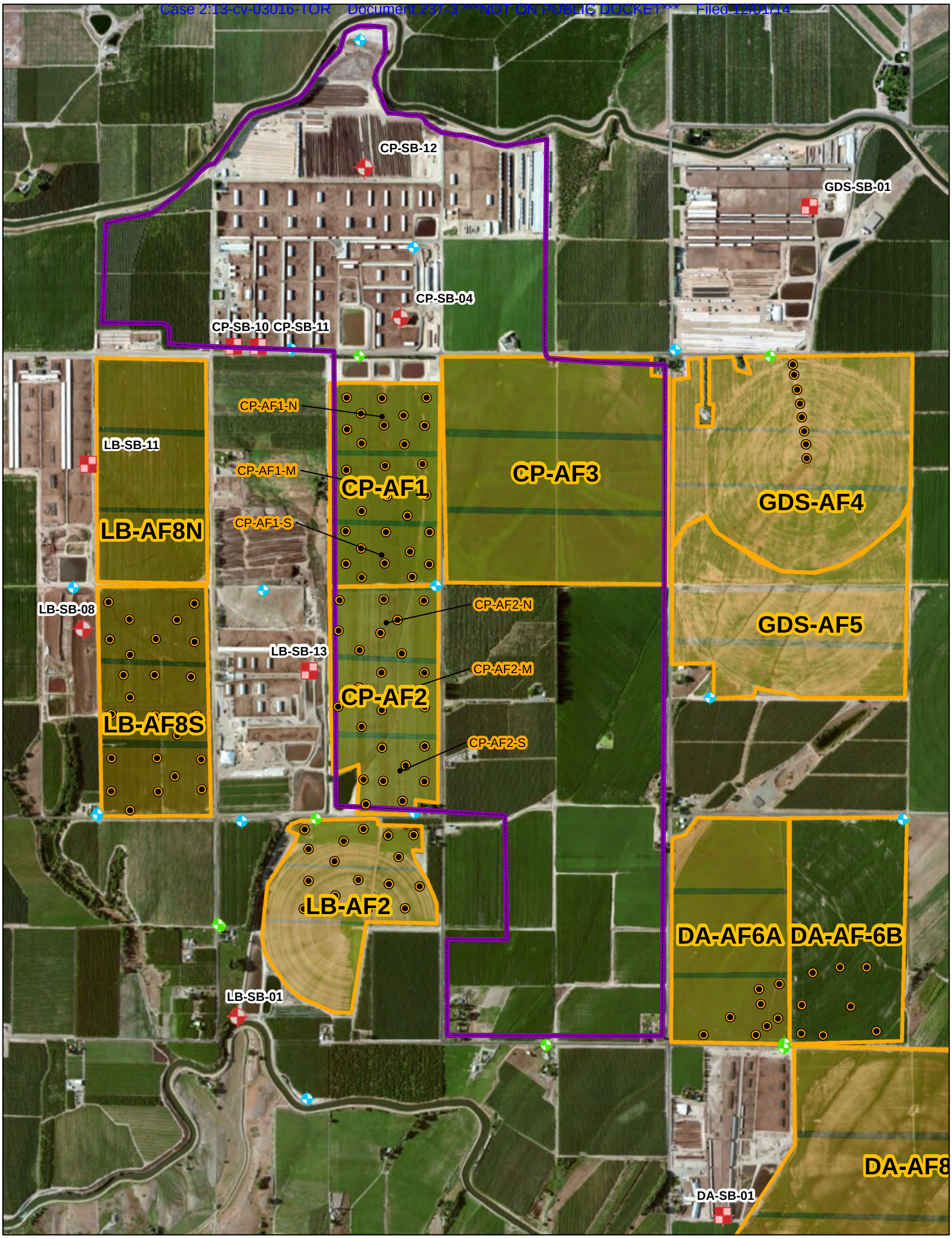


K (cm/s)	w (ft)	ds (ft)	Seepage (GPY/Acre)	Seepage mm/day
1.E-05	12	0.25	165,134,414	423.4
		0.5	84,252,252	216.0
		0.75	57,291,531	146.9
		1	43,811,171	112.3
	8	0.25	111,212,973	285.1
		0.5	57,291,531	146.9
		0.75	39,317,718	100.8
		1	30,330,811	77.8
	4	0.25	57,291,531	146.9
		0.5	30,330,811	77.8
		0.75	21,343,904	54.7
		1	16,850,450	43.2
	2	0.25	30,330,811	77.8
		0.5	16,850,450	43.2
		0.75	12,356,997	31.7
		1	10,110,270	25.9

K (cm/s)	w (ft)	ds (ft)	Seepage GPY/Acre	Seepage mm/day
1.E-06	12	0.25	16,513,441	42.3
		0.5	8,425,225	21.6
		0.75	5,729,153	14.7
		1	4,381,117	11.2
	8	0.25	11,121,297	28.5
		0.5	5,729,153	14.7
		0.75	3,931,772	10.1
		1	3,033,081	7.8
	4	0.25	5,729,153	14.7
		0.5	3,033,081	7.8
		0.75	2,134,390	5.5
		1	1,685,045	4.3
	2	0.25	3,033,081	7.8
		0.5	1,685,045	4.3
		0.75	1,235,700	3.2
		1	1,011,027	2.6

K (cm/s)	w (ft)	ds (ft)	Seepage GPY/Acre	Seepage mm/day
1.E-07	12	0.25	1,651,344	4.23
		0.5	842,523	2.16
		0.75	572,915	1.47
		1	438,112	1.12
	8	0.25	1,112,130	2.85
		0.5	572,915	1.47
		0.75	393,177	1.01
		1	303,308	0.78
	4	0.25	572,915	1.47
		0.5	303,308	0.78
		0.75	213,439	0.55
		1	168,505	0.43
	2	0.25	303,308	0.78
		0.5	168,505	0.43
		0.75	123,570	0.32
		1	101,103	0.26





Legend

- Cow Palace Dairy
- Application Field
- Application Field Composite Area
- Application Field Boring
- Lagoon/Compost/Silage Boring
- Pen Boring
- EPA Well
- Dairy Well

1 Inch = 1500 Feet

0 750 1,500 3,000

N W E S

WATER & ENVIRONMENTAL TECHNOLOGIES, PC

May 2014 Inspection Sample Locations

COW PALACE DAIRY

Job#: CAFOM01

Date: 9/18/2014

FIGURE 1

Path: M:\COW_Palace\May 2014 Inspection Sample Locations.mxd

Cow Palace Dairy, Yakima County, Washington

Surface Water and Lagoon Sample Analytical Results, mg/L

Sample ID	Lab	Date	pH	Solids, Total Suspended TSS @ 105 C	Solids, Total Dissolved TDS @ 180 C	Alkalinity, Total as CaCO3	Chloride	Sulfate	Nitrogen, Ammonia as N	Nitrogen, Kjeldahl, Total as N	Nitrogen, Nitrate as N	Nitrogen, Nitrate + Nitrite as N	Phosphorus, after H2SO4 Hydrolysis	Nitrogen, Nitrite as N	Phosphorus, Orthophosphate as P	Phosphorus, Total as P	Phosphorus, Organic	Phosphorus, Inorganic	Calcium	Magnesium	Potassium	Sodium
Surface Water and Lagoon Contents																						
CP-Catch Basin-SW	EL	10/30/2013	7.3 H	248	236	180	11	5	3.2 D	5.9	0.34	0.47	3.9 D	0.13	1.28 H	4.51	0.63	3.88	38	12.00	24	12
CP-Knowles Rd-SW	EL	10/30/2013	8.3 H	184	1180	1300	98 D	21	90 D	126 D	< 0.01	0.1 D	12 D	0.2	6.42 D	9.9 D	< 0.01	9.9	104	51.00	257	107
CP-R Canal-SW	EL	10/30/2013	8.9 H	19	87	68	4	3	< 0.05	< 0.5	< 0.01	< 0.01	0.03	< 0.01	0.013	0.1	0.07	0.03	14	6.00	1	6
CP-Calf Pen-SW	EL	10/30/2013	7 H	390	350	200	15	21	1.46	13.7	0.4	0.74	3.6	0.33	2.74	4.18	0.54	3.64	44	15.00	20	27
CP-Silage-SW	EL	10/30/2013	3.9 H	590	50100	< 4	990 D	420 D	574 D	2820 D	29.5	29.9 D	966 D	0.37	780 D	898 D	< 0.01	898	1210 D	796.00 D	5090	65
CP-U Storm W-SW	EL	10/30/2013	8.1 H	188	2810	1500	250 D	110 D	61 D	120 D	< 0.01	0.1 D	39 D	0.25	30.3 D	38.2 D	< 0.01	38.2	100	68.00	618	243
CP-L2-SW	EL	10/30/2013	7.6 H	48500	2400	3100	230 D	27 D	330 D	1600 D	< 0.01	0.4 D	358 D	5.9 D	256 D	384 D	26	358	122	49.00	80	26

Notes:
EL indicates Energy Laboratories, Helena, Montana. CAS indicates Cascade Analytical, Wenatchee, WA
H indicates analysis performed past recommended holding time
D indicates reporting limit increased due to sample matrix
J indicates estimated value
-- indicates Not Analyzed

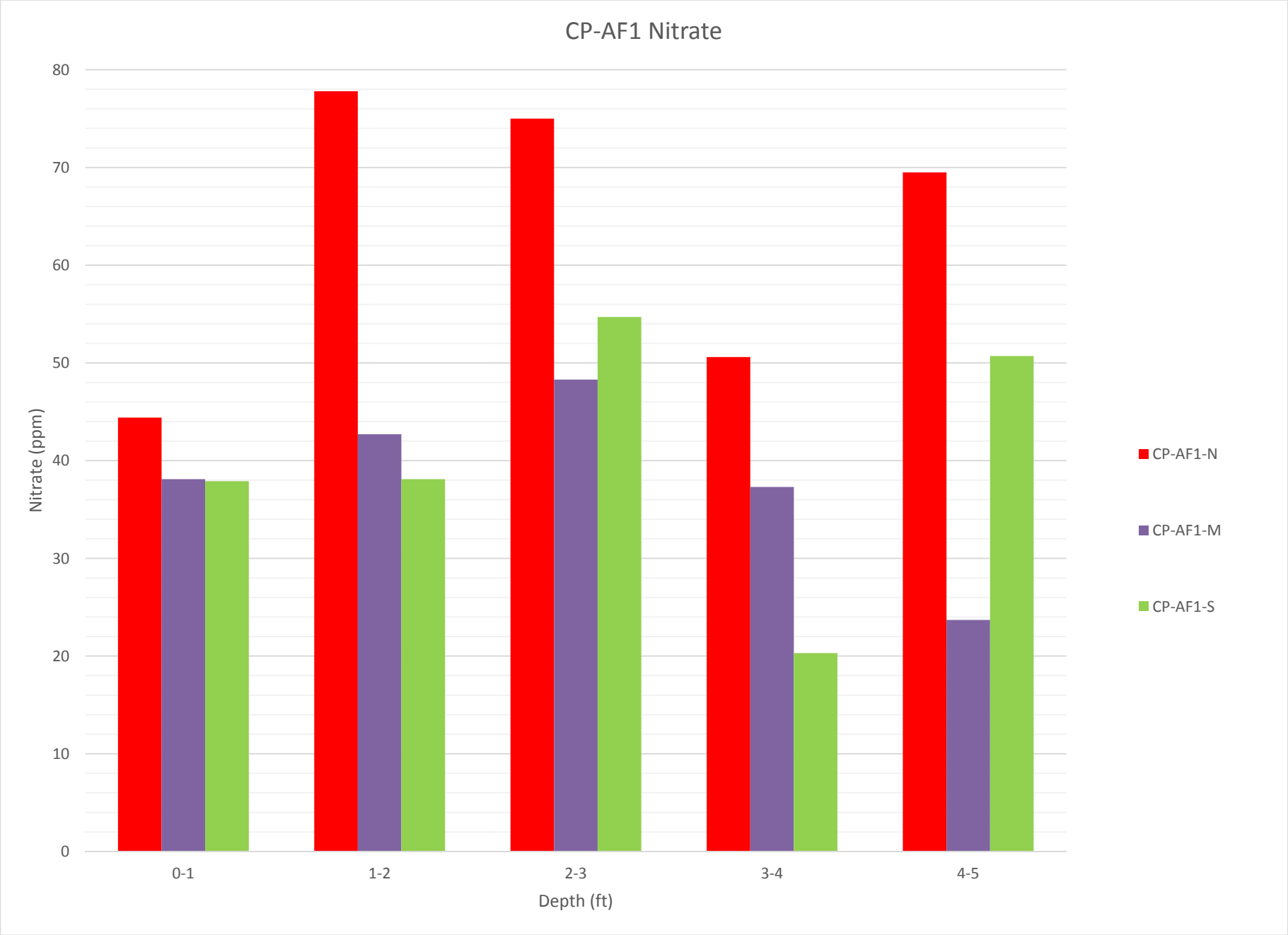
SOIL SAMPLE ANALYTICAL RESULTS

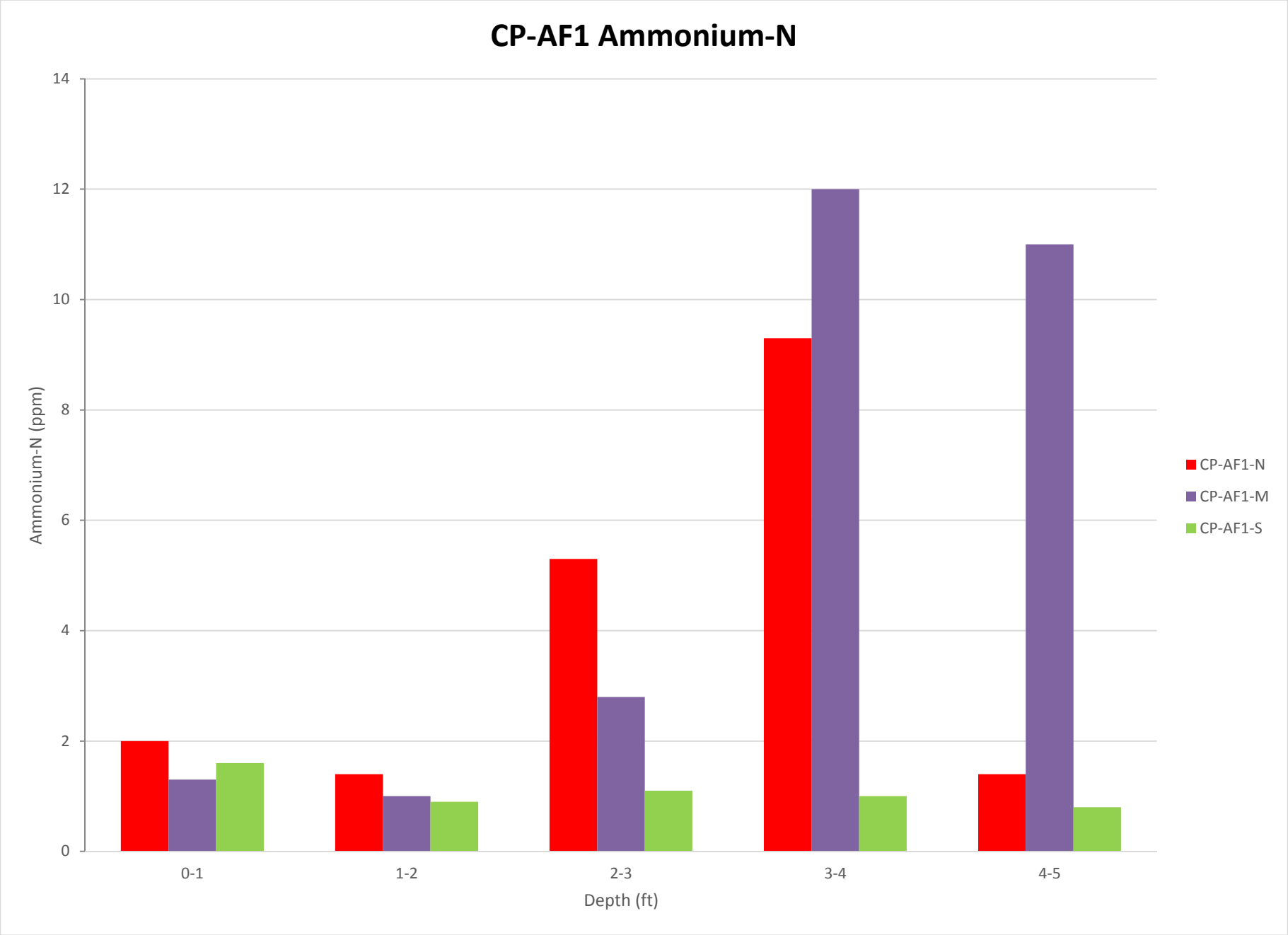
Sample ID	Sample Date	Depth	pH, SU	Phosphorus, ppm	Nitrate, ppm	Ammonium-N, ppm	Total Nitrogen/Solid, mg/kg
COW PALACE							
Application Fields							
CP-AF1-N-0-1	5/19/2014	0-1	8	291	44.4	2	1630
CP-AF1-N-1-2	5/19/2014	1-2	8.3	207	77.8	1.4 J	1150
CP-AF1-N-2-3	5/19/2014	2-3	8.2	118	75	5.3	599
CP-AF1-N-3-4	5/19/2014	3-4	8.3	64.2	50.6	9.3	334
CP-AF1-N-4-5	5/19/2014	4-5	8.3	34.9	69.5	1.4	254
1-CP-AF1-N Grab 3-5ft	5/19/2014	3-5	8	60.6	137	2.2	407
10-CP-AF1-N Grab 3-5ft	5/19/2014	3-5	8.5	45.3	62.3	3.2	233
CP-AF1-M-0-1	5/20/2014	0-1	7.7	352	38.1	1.3	1850
CP-AF1-M-1-2	5/20/2014	1-2	8.1	177	42.7	1	661
CP-AF1-M-2-3	5/20/2014	2-3	8.1	78	48.3	2.8	380
CP-AF1-M-3-4	5/20/2014	3-4	8.2	64.7	37.3	12	308
CP-AF1-M-4-5	5/20/2014	4-5	8.2	40.7	23.7	11	298
8-CP-AF1-M Grab 2-4ft	5/20/2014	2-4	8.2	46.4	48.4	2.8	264
CP-AF1-S-0-1	5/20/2014	0-1	7.8	214	37.9	1.6	1490
CP-AF1-S-1-2	5/20/2014	1-2	8.1	82.6	38.1	0.9	543
CP-AF1-S-2-3	5/20/2014	2-3	8	64.7	54.7	1.1	404
CP-AF1-S-3-4	5/20/2014	3-4	7.8	28.4	20.3	1	251
CP-AF1-S-4-5	5/20/2014	4-5	8.3	41.1	50.7	0.8	165
3-CP-AF1-S Grab 3-5ft	5/20/2014	3-5	8.4	15.4	28.3	0.5	119
5-CP-AF1-S Grab 3-5ft	5/20/2014	3-5	8.4	45.7	38.2	0.6	336
9-CP-AF1-S Grab 3-5ft	5/20/2014	3-5	8	66.6	2.2	36	795
CP-AF2-N-0-1	5/20/2014	0-1	7.9	193	45.7	1.5	1350
CP-AF2-N-1-2	5/20/2014	1-2	8.1	52.3	67.9	0.6	270
CP-AF2-N-2-3	5/20/2014	2-3	7.9	35.8	57.1	0.6	291
CP-AF2-N-3-4	5/20/2014	3-4	7.9	21.8	51.8	< 0.4	238
CP-AF2-N-4-5	5/20/2014	4-5	7.8	18.4	44.7	0.4	274
CP-AF2-M-0-1	5/20/2014	0-1	7.9	173	57.2	1	1230
CP-AF2-M-1-2	5/20/2014	1-2	8.2	42.4	46.6	1.2	237
CP-AF2-M-2-3	5/20/2014	2-3	8.1	29.2	45.3	0.7	< 100
CP-AF2-M-3-4	5/20/2014	3-4	7.8	19.8	49.3	1.2	< 100
CP-AF2-M-4-5	5/20/2014	4-5	7.7	7.9	47.7	1	< 100
CP-AF2-S-0-1	5/20/2014	0-1	7.7	190	24.5	3.2	1430
CP-AF2-S-1-2	5/20/2014	1-2	7.9	69.9	25	0.8	368
CP-AF2-S-2-3	5/20/2014	2-3	7.8	29.8	15.6	0.8	179
CP-AF2-S-3-4	5/20/2014	3-4	8.4	62.8	42	0.6	128
CP-AF2-S-4-5	5/20/2014	4-5	7.9	16.5	28	1	< 100
Storm Water Lagoon Borings							
CP-SB-04C-8-10	5/22/2014	8-10	7.7	38	20.3	1.1	270
CP-SB-04C-10-12	5/22/2014	10-12	7.7	5.1	18.2	0.9	887
CP-SB-04C-13-15	5/22/2014	13-15	7.8	4.9	14.4	0.8	< 100
CP-SB-04C-15-16	5/22/2014	15-16	7.7	5.9	27	1.2	138
CP-SB-04-17.8-18.2	5/19/2014	17.8-18.2	7.2	10.7	22	4.4	112
CP-SB-04-19.5-20	5/19/2014	19.5-20	8	< 1.4	2.9	2	< 100
CP-SB-04C-20-23	5/22/2014	20-23	7.8	< 1.4	7.8	0.5	< 100
CP-SB-04C-27-30	5/22/2014	27-30	7.6	2.1	6.1	0.6	< 100
CP-SB-04C-45.5-47	5/22/2014	45.5-47	7.8	< 1.4	1.2	7.5	< 100

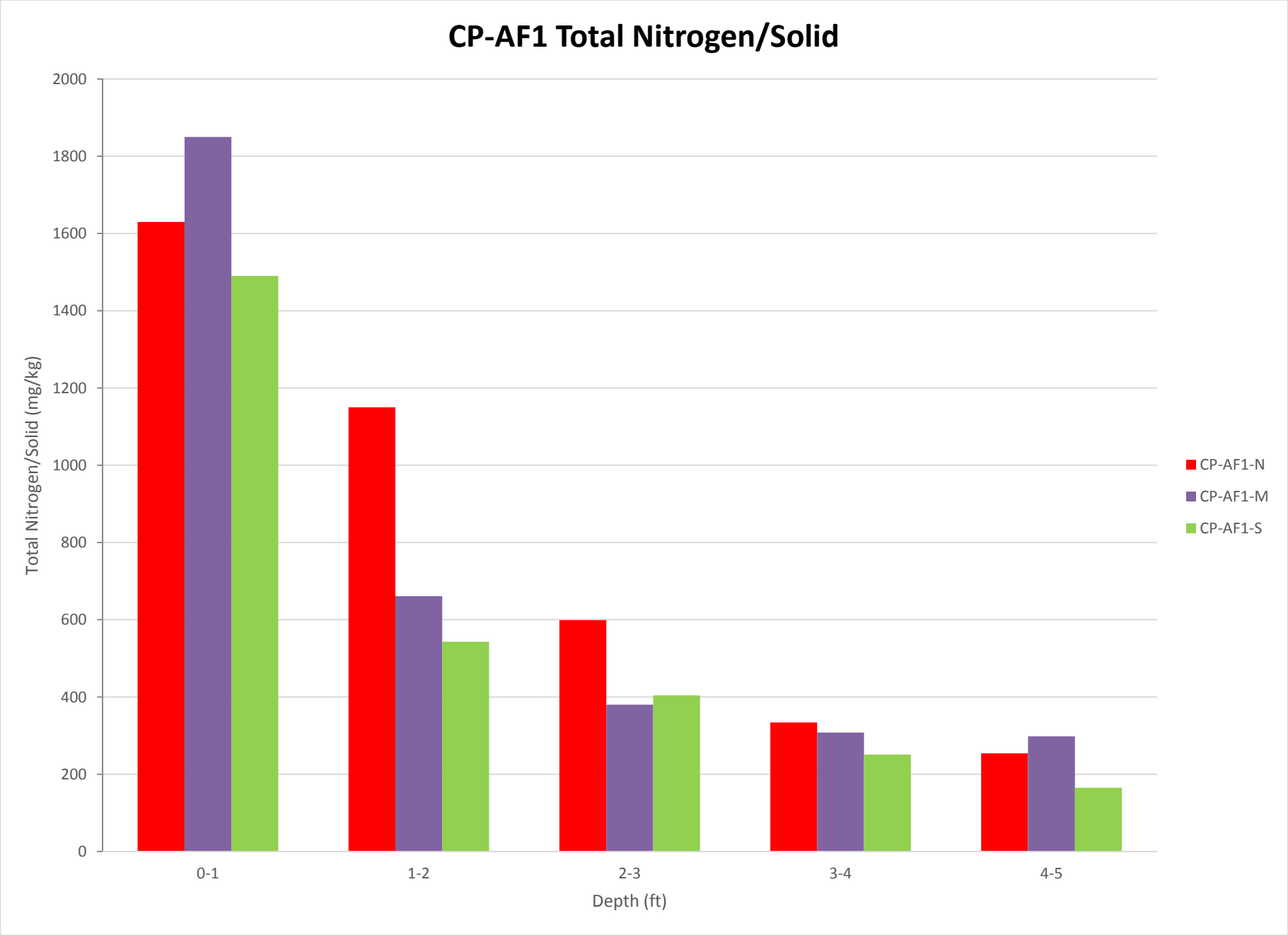
SOIL SAMPLE ANALYTICAL RESULTS

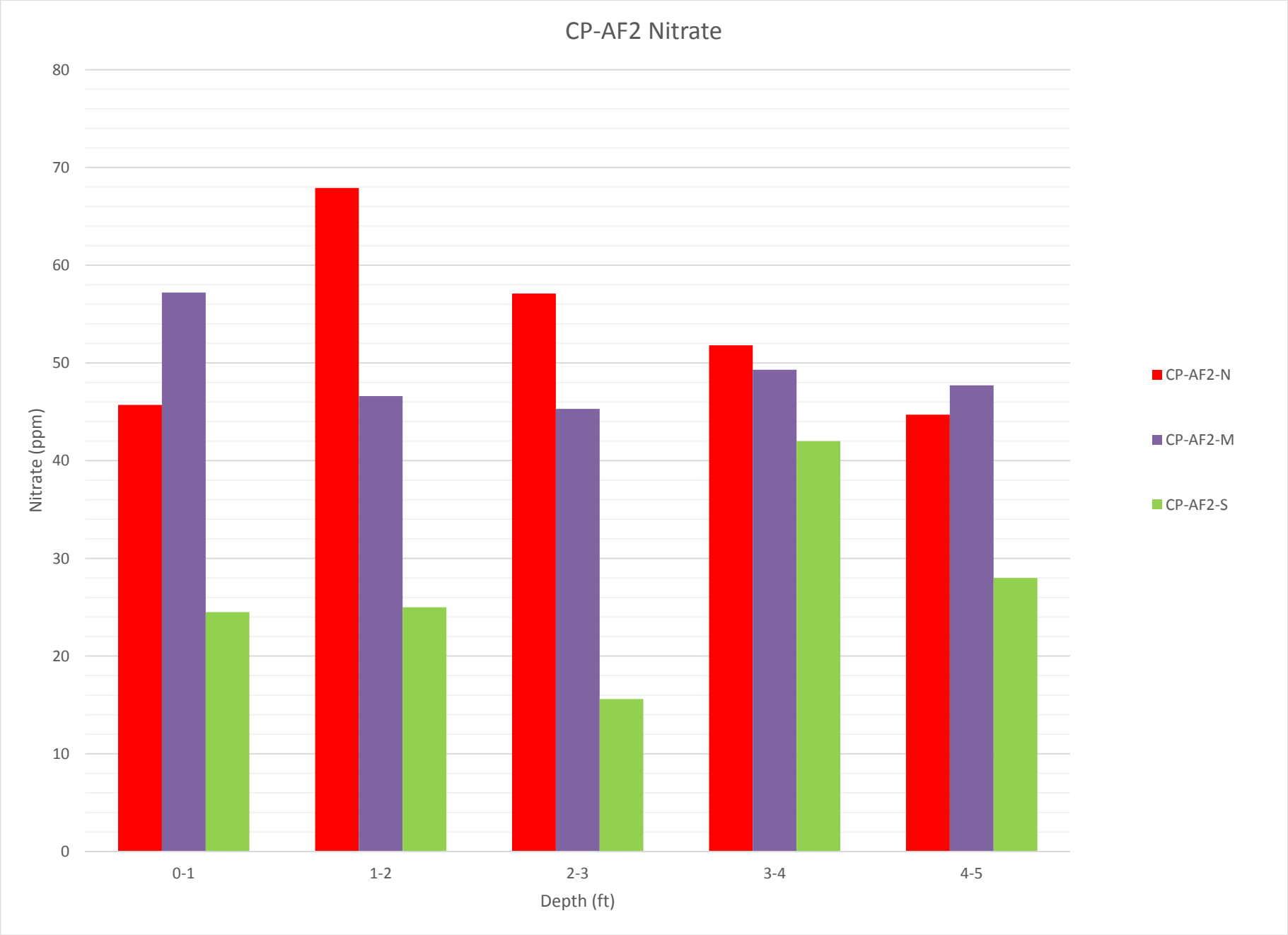
Sample ID	Sample Date	Depth	pH, SU	Phosphorus, ppm	Nitrate, ppm	Ammonium-N, ppm	Total Nitrogen/Solid, mg/kg
Pen Borings							
CP-SB-10-0-1	5/19/2014	0-1	8.2	82	29.9	60	1060
CP-SB-10-1-2	5/19/2014	1-2	7.8	6.5	94.9	8.5	470
CP-SB-10-2-3	5/19/2014	2-3	7.6	5.5	92.1	0.8	295
CP-SB-10-3-4	5/19/2014	3-4	7.9	18.2	40	1.8	358
CP-SB-10-4-5	5/19/2014	4-5	7.9	9.1	8.5	2.4	153
CP-SB-10-5-6	5/19/2014	5-6	8.2	1.5	4.8	3.4	106
CP-SB-10-6-7	5/19/2014	6-7	8.4	1.9	4.7	2.4	126
CP-SB-10-7-8	5/19/2014	7-8	8.5	3.1	2.9	7.1	161
CP-SB-10-9-10	5/19/2014	9-10	8.5	6.5	5.5	2.2	128
CP-SB-11-0-1	5/20/2014	0-1	7.9	39.2	1.9	29	676
CP-SB-11-1-2	5/20/2014	1-2	8.1	75	1.6	160	1090
CP-SB-11-2-3	5/20/2014	2-3	8.7	25.4	14.2	130	591
Compost Boring							
CP-SB-12-0-1	5/19/2014	0-1	8.9	330	12.3	100	2170
CP-SB-12-1-2	5/19/2014	1-2	8	270	5.5	70	1680
CP-SB-12-2-3	5/19/2014	2-3	7.6	51.6	1	20	869
CP-SB-12-3-4	5/19/2014	3-4	7.6	59.4	0.9	14	8210
CP-SB-12-4-5	5/19/2014	4-5	7.5	35.3	49.6	4.5	602
CP-SB-12-5-6	5/19/2014	5-6	7.7	20.2	1.6	12	450
CP-SB-12-6-7	5/19/2014	6-7	7.7	26.4	1	100	818
CP-SB-12-7-8	5/19/2014	7-8	8.6	462	0.9	95	2600
CP-SB-12-8-9	5/19/2014	8-9	8.7	1970	6.8	180	5720
CP-SB-12-10-11	5/19/2014	10-11	8	161	1.6	83	1930
CP-SB-12-11-12	5/19/2014	11-12	8.2	65.2	4.2	19	832
CP-SB-12-12-13	5/19/2014	12-13	7.6	5.1	8.4	5.9	276
CP-SB-12-15-16	5/19/2014	15-16	8.1	7.2	5.1	5.2	133
CP-SB-12-16-17	5/19/2014	16-17	7.9	2.9	2.1	3.5	< 100
CP-SB-12-17-18	5/19/2014	17-18	7.8	1.5	4.3	2.5	< 100

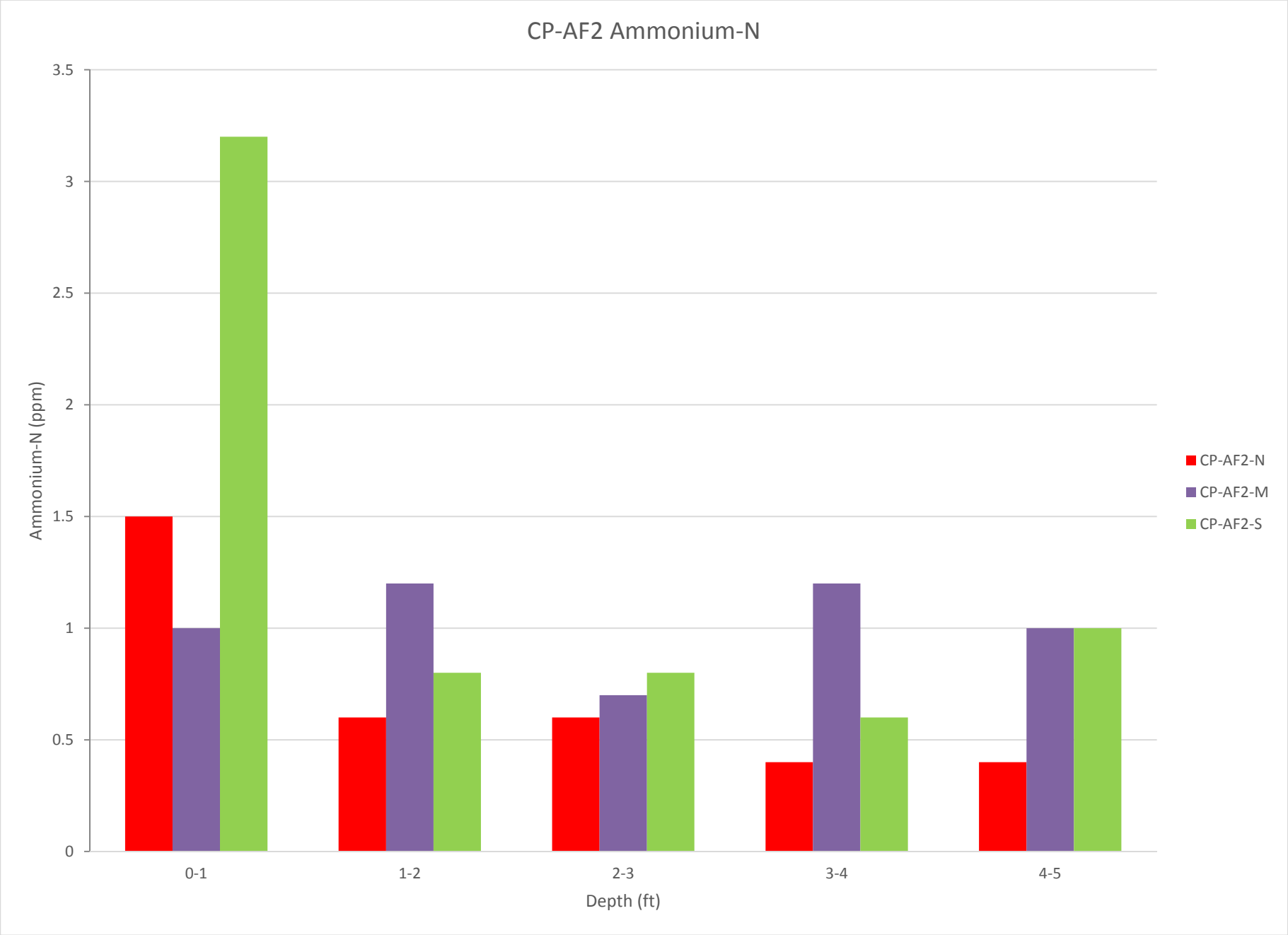
Notes:
J indicates estimated value
< indicates analyte not detected at indicated reporting limit

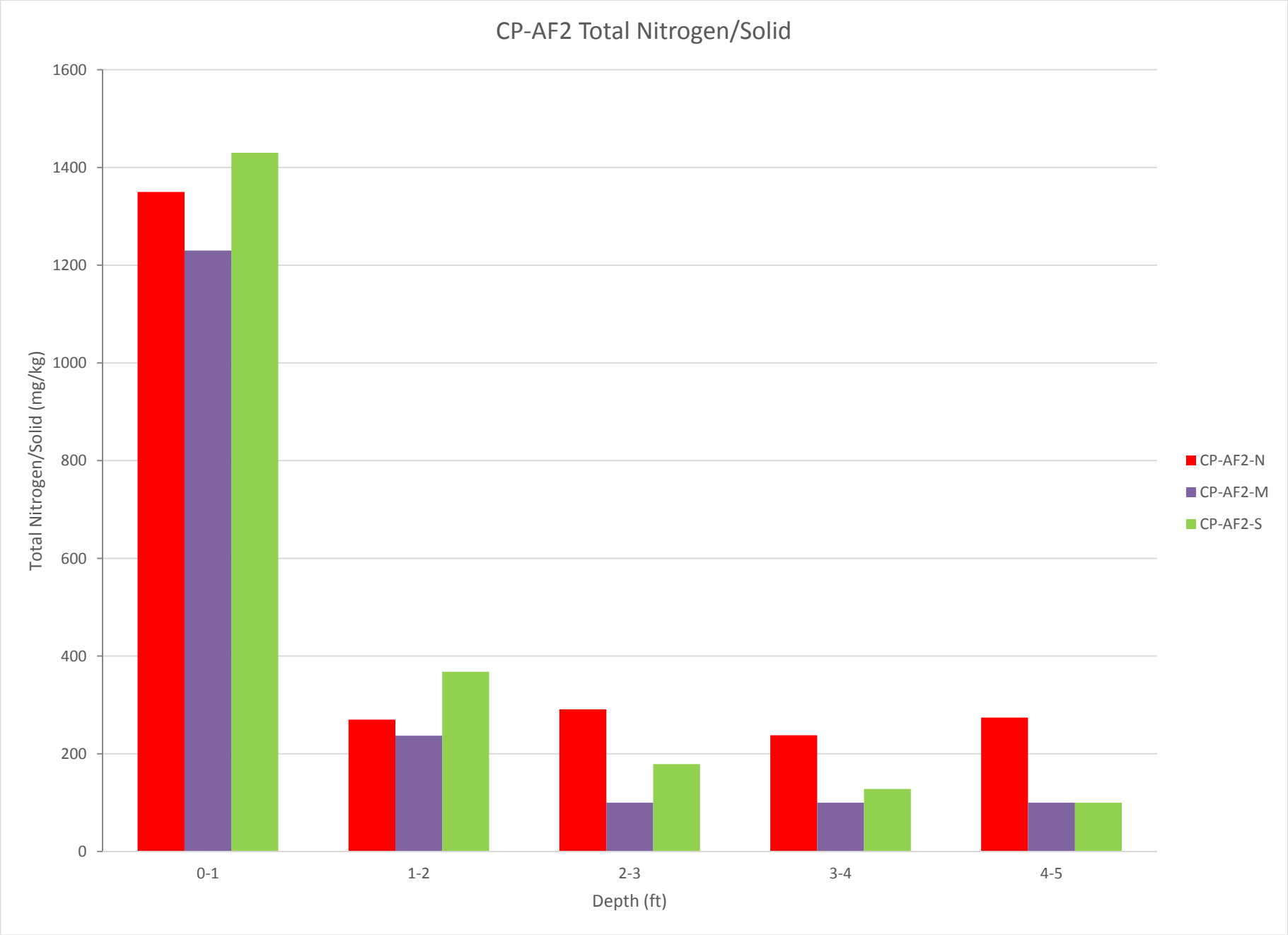

CP-AF1 Nitrate
Nitrate (ppm)
80
70
60
50
40
30
20
10
0
0-1
1-2
2-3
3-4
4-5
Depth (ft)
CP-AF1-N
CP-AF1-M
CP-AF1-S


CP-AF1 Ammonium-N
Ammonium-N (ppm)
14
12
10
8
6
4
2
0
0-1
1-2
2-3
3-4
4-5
Depth (ft)
CP-AF1-N
CP-AF1-M
CP-AF1-S


CP-AF1 Total Nitrogen/Solid
Total Nitrogen/Solid (mg/kg)
2000
1800
1600
1400
1200
1000
800
600
400
200
0
0-1
1-2
2-3
3-4
4-5
Depth (ft)
CP-AF1-N
CP-AF1-M
CP-AF1-S


CP-AF2 Nitrate
Nitrate (ppm)
80
70
60
50
40
30
20
10
0
0-1
1-2
2-3
3-4
4-5
Depth (ft)
CP-AF2-N
CP-AF2-M
CP-AF2-S


CP-AF2 Ammonium-N
Ammonium-N (ppm)
3.5
3
2.5
2
1.5
1
0.5
0
0-1
1-2
2-3
3-4
4-5
Depth (ft)
CP-AF2-N
CP-AF2-M
CP-AF2-S


CP-AF2 Total Nitrogen/Solid
Total Nitrogen/Solid (mg/kg)
1600
1400
1200
1000
800
600
400
200
0
0-1
1-2
2-3
3-4
4-5
Depth (ft)
CP-AF2-N
CP-AF2-M
CP-AF2-S

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From: Cook, Kirk (AGR)
To: Vern Redifer
Cc: Hancock, Jaclyn (AGR)
Subject: N Assessment Update
Date: Friday, May 01, 2015 4:57:42 PM

We have been very busy working on the Livestock portion of the report. Thought you might like some preliminary numbers:

Within the GWMA it looks like we have 212 livestock lagoons....this includes all animal operations not just dairy. This amounts to a capacity of 75,667,000 cu-ft assuming an average lagoon depth of 10 feet. Using the UC Davis report as a starting point we estimate that 54 tons of N is leached to the groundwater every year.

Within the GWMA we have 95 operations with some degree of animal pens (corrals). This amounts to a surface area of 1841.4 acres. Again using the UC Davis report as a starting point we estimate that 824 tons of N is leached to the groundwater every year from corrals.

Within the GWMA we have 24 compost operations of significant size. This amounts to a surface area of 346.5 acres. Using the UC Davis as a starting point we estimate 155 tons of N leached to groundwater a year from these operations. Our records indicate that the vast majority of these operations are conducted directly on the ground and not on a liner or concrete pad.

We will of course be looking at the soil properties within the study area of the UC Davis report and compare them to the soils within the GWMA, this is likely to move the needle a bit (I do not believe that there will be a significant difference in the soil properties).

I understand that the CAFO work group is looking at hiring a consultant to advise of leaching rates at livestock operations.....don't know if there is a good bang for the buck here. I think we have the necessary data to assess the impact. Remember this is an estimate and we have lots of good peer reviewed data to draw upon. Case in point.....the estimates I just provided.

I am in Chicago until Thursday, but given the progress made on the livestock estimates, maybe a meeting of the worker bees is in order soon.

Northwest
Eugene, Oregon

Southwest
Taos, New Mexico

Northern Rockies
Helena, Montana

Southern Rockies
Durango, Colorado

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Via Electronic Mail

March 31, 2014 & April 14, 2014

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Re: Revised Comments on WSDA's Rulemaking to Re-examine Numeric Ranges in WAC 16-16-210(29)

Dear Ms. Prest,

I am submitting a revised version of the following comments that were previously submitted to you on March 31, 2014. The comments are substantively identical to the version of the comments that we submitted on March 31, but the comments are being submitted on behalf of one more organization, Socially Responsible Agriculture Project.

Puget Soundkeeper Alliance (Soundkeeper), the Center for Environmental Law & Policy (CELP), Community Association for the Restoration of the Environment (CARE), Orca Conservancy, Concerned Citizens of the Yakama Reservation (CCYR), Friends of Toppenish Creek, the Water & Salmon Committee of the Washington State Chapter of the Sierra Club, and the Socially Responsible Agriculture Project (SRAP) (collectively the Commenters) appreciate the opportunity to review and provide comment on the Washington Department of Agriculture's ("DOAs") rulemaking to revise the numeric ranges set forth in WAC 16-16-210(29). The Commenters are committed to ensuring that the waters of this state (ground and surface waters) are swimmable, fishable, and sufficiently clean for public consumption. In addition, the Commenters believe that the public has a right to the information they need to protect their

interests in a clean and healthy environment for present and future generations and to ensure that government is fulfilling their obligations to protect the public interest.

The Commenters believe DOA has an opportunity to correct the deficiencies in the existing regulation to bring it into compliance with the language and spirit of the law. In order to fulfill the purposes of the Washington Public Records Act, it is imperative that DOA provide information regarding unpermitted dairies, AFOs and CAFOs that is meaningful to the public. The existing numeric ranges in the regulation are so broad as to be useless and therefore violate the law.

I. INTRODUCTION TO THE COMMENTERS

Puget Soundkeeper Alliance (Soundkeeper) is a 501(c)(3) non-profit environmental organization with a mission of protecting and preserving Puget Sound by monitoring, cleaning up and preventing pollutants from entering its waters. Founded in 1984, Soundkeeper was a founding member of the international Waterkeeper Alliance and today has approximately 3000 members, supporters and volunteers who use and enjoy Puget Sound's marine waters and freshwater tributaries, for commercial, general recreational and aesthetic purposes. To accomplish its mission, Soundkeeper actively monitors Puget Sound through weekly on-water patrols, engages with government agencies and businesses working to regulate pollution discharges, monitors Clean Water Act compliance and enforces the Clean Water Act. Soundkeeper also engages in pollution prevention, including public outreach, cleanup events, citizen advocacy, and technical assistance including the statewide Clean Marina Washington program. With a focus on water quality, Soundkeeper is actively engaged in the public dialogue currently underway relating to discharges from agricultural operations.

The Center for Environmental Law and Policy (CELP) is a 501(c)(3) non-profit whose mission is to dedicated to protect and restore clean, flowing rivers and drinking water aquifers in Washington State through science-based management. CELP is Washington’s Water Watchdog. CELP brings its water law expertise and passion to help citizen groups and tribes throughout

Washington to protect their rivers, streams and aquifers. Since 1993, CELP has advocated for science based, sustainable water management in the legislature, in the courts and with government agencies. CELP's recent accomplishments include establishing that the state law requires protecting a river's aesthetics; convincing the state to issue guidance to counties on how to make land use decisions that impact scarce water resources; and bringing together Columbia River Tribes and leading environmental groups to successfully urge the federal government to revise the Columbia River Treaty to restore the river's beleaguered ecosystem.

The Community Association for the Restoration of the Environment (CARE) is a 501(c)(3) non-profit grassroots organization based in Washington State that is composed of concerned community members. Its mission is to inform Washington State residents about activities that endanger the health, welfare, and quality of life for current and future Washingtonians through education and citizen empowerment.

Orca Conservancy is an all-volunteer, non-profit organization working on behalf of Orcinus species, the killer whales, and protecting the wild places on which they depend. Orca Conservancy collaborates with some of the world's top research institutions and environmental groups to address the most critical issues now facing wild killer whales.

The Concerned Citizens of the Yakima Reservation (CCYR) is an organization made up of life-long residents in the little town of Harrah, WA, (600 people) on the Yakama Indian Reservation, located partially in Yakima County, Washington State. Ten years ago, the community started noticing large amounts of flies in and near their homes, not only in the summer but into the winter months as well. The stench from CAFOs entered their homes, destroying their hope of enjoying a healthy life. Contaminated wells were found along with people having frequent, diarrhea, head colds, sinus problems, and in some cases asthma. Bovine e-coli was found in an air filter of a house. The CCYR was formed to educate the public about the dangers of CAFOs not only in Yakima County and on the Yakama Indian Reservation, but across the nation.

The Friends of Toppenish Creek (FOTC) is a 501(c)(3) nonprofit organization dedicated to protecting the rights of rural communities and improving oversight of industrial agriculture. FOTC operates under the simple principle that all people deserve clean air, clean water and protection from abuse that results when profit is favored over people. FOTC works through public education, citizen investigations, research, legislation, special events, and direct action.

The Water & Salmon Committee, Washington State Chapter of the Sierra Club is a part of the national environmental organization whose mission is to explore, enjoy and protect the planet; to practice and promote the responsible use of the earth's ecosystems and resources; to educate and enlist humanity to protect and restore the quality of the natural and human environment; and to use all lawful means to carry out these objectives.

Socially Responsible Agricultural Project ("SRAP") is a grassroots organization that educates the public about the devastating effects of CAFOs, while working directly with the communities most heavily impacted by these animal factories. Through education, advocacy, and community organizing, SRAP empowers rural communities to protect themselves from CAFOs and provides guidance and assistance to communities seeking to develop healthy, sustainable alternatives to industrialized livestock production.

II. LEGAL BACKGROUND

A. The WA Public Records Act Guarantees Citizen Rights To Public Information

WAC 16-16-210(29) is DOA's failed attempt to implement two separate, but nearly identical statutory sections, RCW 42.56.610 and RCW 90.64.190.[1] The first, RCW 42.56.610 is

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[1] RCW 90.64.190 is within Washington's Dairy Nutrient Management Act and is in all material respects identical to RCW 42.56.610:

> This section applies to dairies, AFOs, and CAFOs, not required to apply for a permit. Information in plans, records, and reports obtained by state and local agencies from livestock producers under chapter 510, Laws of 2005 regarding (1) number of animals; (2) volume of livestock nutrients generated; (3) number

within Washington's Public Records Act. As a starting point, it is important to recognize the breadth of the public's right to information guaranteed by the Public Records Act:

> The people of this state do not yield their sovereignty to the agencies that serve them. The people, in delegating authority, do not give their public servants the right to decide what is good for the people to know and what is not good for them to know. The people insist on remaining informed so that they may maintain control over the instruments that they have created. This chapter shall be liberally construed and its exemptions narrowly construed to promote this public policy and to assure that the public interest will be fully protected. In the event of conflict between the provisions of this chapter and any other act, the provisions of this chapter shall govern.

RCW 42.56.030. In fulfilling this objective, state agencies are "to provide full public access to public records" and "to protect public records from damage or disorganization." *See Sanders v. State*, 169 Wn.2d 827, 846, 240 P.2d P.3d 120 (2010) (stating that "we have consistently enforced the PRA's disclosure requirements to advance its policy of public access."); *Neighborhood Alliance of Spokane Cnty. v. Cnty. of Spokane*, 172 Wash.2d 702, 714, 261 P.3d 119 (2011) ("The PRA is a strongly worded mandate for broad disclosure of public records.").

The Public Records Act, however, does create certain, narrow exemptions to the disclosure of public records. The case law is clear that these exemptions are to be narrowly construed. *Gale v. City of Seattle*, 2014 WL 545844 *6 (Wash. Ct. App. Feb. 10, 2014) (unpublished decision) ("We liberally construe the PRA in favor of disclosure and narrowly construe its exemptions.") (citing RCW 42.56.030). In addition, "a court may enjoin production of requested records if an exemption applies and *examination would clearly not be in the public*

> of acres covered by the plan or used for land application of livestock nutrients; (4) livestock nutrients transferred to other persons; and (5) crop yields shall be disclosable in response to a request for public records under chapter 42.56 RCW only in ranges that provide meaningful information to the public while ensuring confidentiality of business information. The department of agriculture shall adopt rules to implement this section in consultation with affected state and local agencies.

interest and would substantially and irreparably damage any person, or would substantially and irreparably damage vital governmental functions." *Robbins, Geller, Rudman & Dowd, LLP v. State*, ___ P.3d ___, 2014 WL 839895 *2 (Wash. Ct. App.) (March 4, 2014) (emphasis added).

There are categorical exemptions designed to protect an individual's right to privacy or any vital government function, but both of these exemptions are also narrowly applied on a case-specific basis. RCW 42.56.050; RCW 42.56.210(1) ("the exemptions of [the Public Records Act] are inapplicable to the extent that information, the disclosure of which would violate personal privacy or vital governmental interests, can be deleted from the specific records sought. No exemption may be construed to permit the nondisclosure of statistical information not descriptive of any readily identifiable person or persons."). There are a number of fact-specific exemptions regarding, for example, archaeological sites (RCW 42.56.300); crime victim information (RCW 42.56.240); or educational information (RCW 42.56.320).

The legislature has created exemptions specifically related to agriculture and livestock, none of which are applicable to DOA's proposed rulemaking that is the subject of these comments. RCW 42.56.380. The agency invoking the exemption to disclose public records has the burden to establish that a specific exemption applies. *Neighborhood Alliance of Spokane Cnty.*, 172 Wn.2d at 715. RCW 42.56.610, the statutory source of authority for WAC 16.16.210(29), creates another exemption for the agricultural industry in Washington's Public Records Act and states as follows:

> The following information in plans, records, and reports obtained by state and local agencies from dairies, animal feeding operations, and concentrated animal feeding operations, not required to apply for a national pollutant discharge elimination system permit is disclosable only in ranges that provide meaningful information to the public while ensuring confidentiality of business information regarding: (1) Number of animals; (2) volume of livestock nutrients generated; (3) number of acres covered by the plan or used for land application of livestock nutrients; (4) livestock nutrients transferred to other persons; and (5) crop yields. The department of agriculture shall adopt rules to implement this section in consultation with affected state and local agencies.

RCW 42.56.610.[2] This section creates a limited statutory exception to the presumption that all information in plans, records and reports held by the state regarding dairies, animal feeding operations, and concentrated animal feeding operations is subject to public disclosure. RCW 42.56.070(1) ("Each agency . . . *shall* make available for public inspection and copying *all public records*, unless the record falls within the specific exemptions . . . .") (emphasis added). In interpreting and applying this statutory exemption, DOA is required, to define "ranges that provide meaningful information to the public while ensuring confidentiality of business information" for each specified category of information. In doing so, DOA is legally obligated to interpret and apply this exemption narrowly. By adopting the existing WAC 16.16.210(29), DOA has failed to comply with the Public Records Act and case law interpreting this statute.

B. THE CURRENT REGULATION VIOLATES THE LAW

a. The Current Regulation Does Not Provide Meaningful Information

WAC 16-16-210(29) does not provide a definition of what constitutes "meaningful information to the public." Rather, it simply creates wide numerical ranges for each specified category of information. The lack of a definition of "meaningful information" is a fatal flaw that is exacerbated by the fact that the ranges set forth in WAC 16-16-210(29) demonstrably not provide "meaningful information" as discussed in detail below.

Under the Public Records Act, information is "meaningful" if it has the potential to affect the public interest or if it constitutes information relevant to the people's ability to monitor

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[2] It is important to note that this regulation only applies to dairies, AFOs and CAFOs "not required to apply for a national pollutant discharge elimination system [NPDES] permit . . . ." RCW 42.56.610. However, because all CAFOs are point sources under the plain language of the Clean Water Act, and because all CAFOs discharge to waters of the state either by surface water discharges or directly to groundwater, all CAFOs should be required to have NPDES permits. Clean Water Act, Section 502(14).

agency compliance with the law. RCW 42.56.030 ("The people, in delegating authority, do not give their public servants the right to decide what is good for the people to know and what is not good for them to know."); *Harley H. Hoppe & Assoc., Inc. v. King County*, 162 Wash.App. 40, 54-55, 255 P.3d 819 (2011) ("The Public Records Act shall be liberally construed and its exemptions narrowly construed to promote this public policy and *to assure that the public interest will be fully protected*.") (emphasis added). The Legislature has made it very clear that the Public Records Act is designed to protect the public's right to information that has the potential to affect their legally-secured rights.

> [T]he legislature finds that public health and safety is promoted when the public has knowledge that enables them to make informed choices about their health and safety. Therefore, the legislature declares, as a matter of public policy, that the public has a right to information necessary to protect members of the public from harm caused by alleged hazards or threats to the public.

RCW 42.56.210. In 1972, the citizens of this state enacted the Public Disclosure Act, which was later partially re-codified as the existing Public Records Act. *NW Gas Ass'n v. WA Utilities & Transp. Comm'n*, 141 Wash.App. 98, 106-07, 168 P.23d 443 (2007). The stated policy of the Public Disclosure Act was "'to promote . . . full access to public records so as to assure continuing public confidence of fairness of elections and governmental processes, and *so as to assure that the public interest will be fully protected*.'" *Id*. (quoting RCW 42.17.010) (emphasis added). The Public Records Act clearly and concisely dictates a public policy of ensuring that the public "remain[] informed so that they may maintain control over the instruments that they have created." RCW 42.56.030; *see Tobin v. Warden*, 156 Wash.App. 507, 513 n.3, 233 P.3d 906 (2010) ("In construing the PRA, courts look at the act in its entirety to enforce the law's overall purpose."). The information that is automatically exempted from disclosure under DOA's interpretation of RCW 42.56.610 impedes the public's ability to ascertain how unpermitted dairies, AFOs and CAFOs are affecting their rights as citizens of this state. In addition, it impedes the public's ability to ascertain whether state agencies such as Agriculture,

Ecology and Health, are doing what they are legally obligated to do to protect the public from the pollution caused by these facilities.

i. The Public Has An Interest in a Clean & Healthy Environment

Several sources of Washington state law confirm and protect citizens' inherent rights to a clean and healthy environment. Citizen rights in this regard are relevant to this rulemaking proceeding because the existing regulation impedes citizens from exercising their rights to the fullest extent of the law by denying them crucial information. The Washington legislature adopted the State Environmental Policy Act ("SEPA"), recognizing that:

> (2) It is the continuing responsibility of the state of Washington and all agencies of the state to use all practicable means, consistent with other essential considerations of state policy, to improve and coordinate plans, functions, programs, and resources to the end that the state and its citizens may: (a) *Fulfill the responsibilities of each generation as trustee of the environment for succeeding generations* . . . . (3) *The legislature recognizes that each person has a fundamental and inalienable right to a healthful environment and that each person has a responsibility to contribute to the preservation and enhancement of the environment.*

RCW 43.21C.020(2), (3) (emphasis added). The Shoreline Management Act provides that state shoreline management policy "contemplates protecting against adverse effects to the *public health, the land and its vegetation and wildlife, and the waters of the state and their aquatic life*, while protecting generally public rights of navigation and corollary rights incidental thereto." RCW 90.58.020 (emphasis added). In regards to wildlife, the Washington legislature has asserted dominion, and even ownership, over wildlife found within the state, on behalf of state citizens:

> Wildlife, fish, and shellfish are the property of the state. The commission, director, and the department shall preserve, protect, perpetuate, and manage the wildlife and food fish, game fish, and shellfish in state waters and offshore waters. The department shall conserve the wildlife and food fish, game fish, and shellfish resources in a manner that does not impair the resource.

RCW 77.04.012. This exercise of dominion/control carries with it a corollary obligation to ensure that the wildlife, fish and shellfish of this state exist for present and future generations. *See Nelson Alaska Seafoods, Inc. v. Dep't of Revenue,* 143 Wash. App. 455, 462, 177 P.3d 1161 (2008) ("DNR regulates the commercial geoduck harvest for the public good . . . ."); *Lake Union Drydock Co., Inc. v. State Dept. of Natural Resources*, 143 Wash. App. 644, 658, 179 P.3d 844 (2008) (quoting former RCW 79.90.450) ("To implement this public trust, the Legislature expressly delegated authority to the DNR to manage state-owned aquatic lands for "the benefit of the public . . . .'"); *Wash. Geoduck Harvest Ass'n v. Dep't of Natural Res.*, 124 Wash. App. at 449.

The State has unequivocally asserted its sovereign dominion and control over the air resources of this state:

> It is declared to be the public policy to preserve, protect, and enhance the air quality for current and future generations. *Air is an essential resource that must be protected from harmful levels of pollution. Improving air quality is a matter of statewide concern and is in the public interest.* It is the intent of this chapter to secure and maintain levels of air quality that protect human health and safety, including the most sensitive members of the population, to comply with the requirements of the federal clean air act, to prevent injury to plant, animal life, and property, to foster the comfort and convenience of Washington's inhabitants, to promote the economic and social development of the state, and to facilitate the enjoyment of the natural attractions of the state. It is further the intent of this chapter to protect the public welfare, to preserve visibility, to protect scenic, aesthetic, historic, and cultural values, and to prevent air pollution problems that interfere with the enjoyment of life, property, or natural attractions.

RCW 70.94.011 (emphasis added). These statutory provisions demonstrate and affirm the public's inherent right to a clean and healthy environment in the state of Washington. It necessarily follows that the public, then, has a right to information that has the potential to affect that right.

The public's interest in clean water is explicitly clear in this state. The state's water code, RCW 90.03, declares that:

> It is the policy of the state to promote the use of the public waters in a fashion which provides for obtaining maximum net benefits arising from both diversionary uses of the state's public waters and the retention of waters within streams and lakes in sufficient quantity and quality to protect instream and natural values and rights.

RCW 90.03.005. The Legislature has prioritized the maintenance of instream flows in waters of the state "for the purposes of protecting fish, game, birds or other wildlife resources, or recreational or aesthetic values of said public waters whenever it appears to be in the public interest to establish the same." RCW 90.22.010. This statute gives Ecology broad authority to protect instream flows for purposes of both water quality and water quantity. *Id.* The state's Water Pollution Control Act, RCW 90.48 declares the following public policy:

> It is declared to be the public policy of the state of Washington to maintain the highest possible standards to insure the purity of all waters of the state consistent with public health and public enjoyment thereof, the propagation and protection of wild life, birds, game, fish and other aquatic life, and the industrial development of the state, and to that end require the use of all known available and reasonable methods by industries and others to prevent and control the pollution of the waters of the state of Washington. Consistent with this policy, the state of Washington will exercise its powers, as fully and as effectively as possible, to retain and secure high quality for all waters of the state. The state of Washington in recognition of the federal government's interest in the quality of the navigable waters of the United States, of which certain portions thereof are within the jurisdictional limits of this state, proclaims a public policy of working cooperatively with the federal government in a joint effort to extinguish the sources of water quality degradation, while at the same time preserving and vigorously exercising state powers to insure that present and future standards of water quality within the state shall be determined by the citizenry, through and by the efforts of state government, of the state of Washington.

RCW 90.48.010. The Water Resources Act of 1971 declares:

> (3) The quality of the natural environment shall be protected and, where possible, enhanced as follows: (a) Perennial rivers and streams of the state shall be retained with base flows necessary to provide for preservation of wildlife, fish, scenic, aesthetic and other environmental values, and navigational values. Lakes and ponds shall be retained substantially in their natural condition. Withdrawals of water which would conflict therewith shall be authorized only in those situations where it is clear that overriding considerations of the public interest will be served. (b) Waters of the state shall be of high quality. Regardless of the quality of the waters of the state, all wastes and other materials and substances proposed for entry into said waters shall be provided with all known, available, and reasonable methods of treatment prior to entry. Notwithstanding that standards of quality established for the waters of the state would not be violated, wastes and other materials and substances shall not be allowed to enter such waters which will reduce the existing quality thereof, except in those situations where it is clear that overriding considerations of the public interest will be served.

RCW 90.54.020(3). In regards to groundwater,

> The legislature hereby declares that *the protection of groundwater aquifers* which are the sole drinking water source for a given jurisdiction *shall be of the uppermost priority* of the state of ecology, department of social and health services, and all local government agencies with jurisdiction over such areas. In administration of programs related to the disposal of wastes and other practices which may impact water quality, the department of ecology, department of social and health services, and such affected local agencies *shall explore all possible measures for the protection of the aquifer, including any appropriate incentives, penalties, or other measures designed to bring about practices which provide for the least impact on the quality of the groundwater.*

RCW 90.54.140 (emphasis added). Finally, the Dairy Nutrient Management Act also declares a policy of addressing "the discharge of pollution to surface and ground waters of the state that will lead to water quality compliance by the industry." RCW 90.64.005. Therefore, it is quite clear that the public has a strong interest, protected by the plethora of state laws discussed above, in a clean environment in general, and clean water is particular. The public also has a right to ensure that industries that generate pollution are complying with the law. The public

has a clear legal right to access information that has the potential to affect the public interest in any way, RCW 42.56.030, and that right must be safeguarded by WAC 16.06.210(29).

2. **The Public Has A Right To Protect Public Health**

The Legislature has specifically recognized that people in the state have a right to ensure that state water resources are utilized in a way that promotes, and not harms, public health. The Legislature as found:

> (a) *Proper utilization of the water resources of this state is necessary to the promotion of public health and the economic well-being of the state and the preservation of its natural resources and aesthetic values*. Although water is a renewable resource, its supply and availability are becoming increasingly limited, particularly during summer and fall months and dry years when demand is greatest. Growth and prosperity have significantly increased the competition for this limited resource. *Adequate water supplies are essential to meet the needs of the state's growing population and economy. At the same time instream resources and values must be preserved and protected so that future generations can continue to enjoy them*.
> (b) *All citizens of Washington share an interest in the proper stewardship of our invaluable water resources*. To ensure that available water supplies are managed to best meet both instream and offstream needs, a comprehensive planning process is essential. The people of the state have the unique opportunity to work together to plan and manage our water. Through a comprehensive planning process that includes the state, Indian tribes, local governments, and interested parties, it is possible to make better use of available water supplies and achieve better management of water resources. Through comprehensive planning, conflicts among water users and interests can be reduced or resolved. It is in the best interests of the state that comprehensive water resource planning be given a high priority so that water resources and associated values can be utilized and enjoyed today and protected for tomorrow.

RCW 90.54.010(a), (b) (emphasis added). The Washington legislature has explicitly directed the Washington State Board of Health to regulate the storage of animal waste from dairies, AFOs or CAFOs to protect human health:

> In order to protect public health, the state board of health shall: Adopt rules and standards for prevention, control, and abatement of health hazards and nuisances related to the disposal of human and animal excreta and animal remains.

RCW § 43.20.050(2)(c) (2013). The Board's statutory obligation to protect public health from pollutants created by dairies, AFOs and CAFOs is an affirmative delegation of state police power to protect the public's interest in good health. The legislature has directed all local boards of health and health officers, among other state and local officials, to enforce the regulations promulgated by the Board of Health to carry out this paramount duty of protecting the health of Washingtonians. RCW§ 43.20.050(5).

The Department of Health's Office of Drinking Water is the regulatory body charged with ensuring the safety of public drinking water systems in accordance with the Safe Drinking Water Act (SDWA). Pub. L. 93-523. The SDWA mandates the prevention of pollution of public drinking water from any source, including dairies, AFOs and CAFOs.

> To ensure that drinking water is safe, SDWA sets up multiple barriers against pollution. These barriers include: source water protection, treatment, distribution system integrity, and public information. Public water systems are responsible for ensuring that contaminants in tap water do not exceed the standards. Water systems treat the water, and must test their water frequently for specified contaminants and report the results to states. If a water system is not meeting these standards, it is the water supplier's responsibility to notify its customers. Many water suppliers now are also required to prepare annual reports for their customers. The public is responsible for helping local water suppliers to set priorities, make decisions on funding and system improvements, and establish programs to protect drinking water sources. Water systems across the nation rely on citizen advisory committees, rate boards, volunteers, and civic leaders to actively protect this resource in every community in America.[3]

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[3] U.S. EPA, *Understanding the Safe Drinking Water Act, available at* *http://water.epa.gov/lawsregs/guidance/sdwa/upload/2009_08_28_sdwa_fs_30ann_sdwa_web.pdf* (last visited December 3, 2013).

Therefore, the public has a strong interest, protected by the law and moral necessity,[4] in knowing what pollutants that have the potential to affect public health are entering our commonly-shared water and air resources.

3. The Public Has A Right To Know How Farm Animals Are Treated

Disclosure of accurate stocking numbers at dairies, AFOs and CAFOs is crucial to monitoring consumer safety and animal welfare in the beef and dairy industry. Cows suffer generally from crowded conditions, exhibiting, for example, more competitive behavior and shorter lying times, both associated with increased stocking density.[5] Crowding also contributes to the spread of disease, including bovine paratuberculosis, or Johne's disease—an untreatable wasting disease that lacks a vaccine. Inspection manuals for Johne's disease specifically identify stocking density as a risk factor.[6] Bovine respiratory disease is also caused or exacerbated by crowding.[7] The risk of disease is not limited to cattle. Crowded conditions also increase the risk of the spread of bacterial diseases to consumers, including *Escherichia coli* (*E. coli)*.[8] Because

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[4] The moral justification to act to protect our drinking water sources from nitrate pollution is especially strong given the fact that young infants are more susceptible to nitrate contamination than adults. *See* Minnesota Department of Health, "Nitrate in Well Water," *available at* http://www.health.state.mn.us/divs/eh/wells/waterquality/nitrate.html#Whyare (last visited March 31, 2014). If we as adults do not act to protect the young infants in this state, who will? Furthermore, parents of young infants should be able to access the information that they need to fully protect their infant children.

[5] *See* A. Talebi, et al., Reduced stocking density mitigates the negative effects of regrouping in dairy cattle, Journal of Dairy Science, Vol. 97, 1358 (Mar. 2014). *Abstract available* at http://www.journalofdairyscience.org/article/S0022-0302%2814%2900014-9/abstract.

[6] *See* Handbook for Vets & Beef Producers, *available for download at* http://www.johnesdisease.org/.

[7] *See* http://www.bovilis.com/diseases/brdc/etiology.asp.

[8] The largest *E. coli* outbreak in Washington state was in 1993, when 477 people were sickened by contaminated beef. *See* http://www.doh.wa.gov/YouandYourFamily/IllnessandDisease/Ecoli.aspx. *E. coli* is carried in

the public ostensibly consumes the meat and dairy products produced at non-permitted dairies, AFOs and CAFOs in the state, they have a right to accurate stocking numbers.

In regards to chicken AFOs and CAFOs, population densities in factory egg production facilities bear directly on animal welfare in the egg industry. Accurate information on the number of animals within chicken AFOs and CAFOs is meaningful to the public, and thus should be subject to disclosure, because population densities can affect sanitary conditions in these operations and lack of adequate sanitation can directly affect the safety of the egg product produced there.

4. Dairies, AFOs and CAFOs Affect the Public Interest By Polluting Ground & Surface Water

The current state of ground water contamination by agricultural sources in Washington, and the attendant public health risks, is well documented. In a 2001-02 study conducted in the Lower Yakima Valley, "21% of the wells sampled in Region 2 exceeded [the U.S. EPA's MCL for nitrate + nitrate of 10 mg/L]. Mean values for ammonia, chloride, and specific conductivity were also significantly higher in Region 2 . . . . [O]ther studies have shown that overuse of nitrogen fertilizers [including manure] is the primary cause of nitrate contamination of groundwater in agricultural areas."[9] Another study conducted in 2004 determined that "[t]he locations of wells that test positive for total coliforms are in areas of high groundwater [nitrate] concentrations. The source of these bacteria can only be animal feces. Consequently, these results suggest that sources of contaminants are feedlots and/or dairy operations."[10]

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diarrhea, which is spread by overcrowding. *See*, *e.g.*, *veterinaryextension.colostate.edu/menu2/Cattle/Calf%20Scours%20101.pdf at 1 ("Overcrowding is a major contributing factor to calf [diarrhea].").*

[9] *Quality of Ground Water in Private Wells in the Lower Yakima Valley*, Valley Institute for Research & Education (2001-02).

[10] *Sunnyside Groundwater Study Final Report* (Heritage College, 2003).

In 2011, the EPA conducted extensive groundwater monitoring in the Lower Yakima Valley in accordance with the Safe Drinking Water Act. EPA noted that "[n]itrate levels above EPA's drinking water standard in residential drinking wells in the Lower Yakima Valley are well documented."[11] The EPA went on to identify the likely source of the extensive contamination:

> Given the historic and current volumes of wastes generated and stored by dairies, and the application of nitrogen-rich fertilizers including dairy waste in the Lower Yakima Valley, it is expected that dairies are a likely source of of high nitrate levels in downgradient drinking water wells. *The [data] provide strong evidence that the dairies evaluated in this study are likely sources of the high nitrate levels in the drinking water wells downgradient of the dairies.* Additional information that supports this conclusion includes: there are few potential sources of nitrogen located upgradient of the dairies; the dairy lagoons are likely leaking large quantities of nitrogen-rich liquid into the subsurface; and Washington State Department of Agriculture inspectors have reported elevated levels of nitrogen in application fields of the dairies in the study.

Id. (emphasis added). The EPA has identified dairies/CAFOs as the primary source of nitrogen contamination in the Lower Yakima Valley.

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[11] *Relation Between Nitrate in Water Wells and Potential Sources in the Lower Yakima Valley, Washington* (EPA-910-R-12-003).

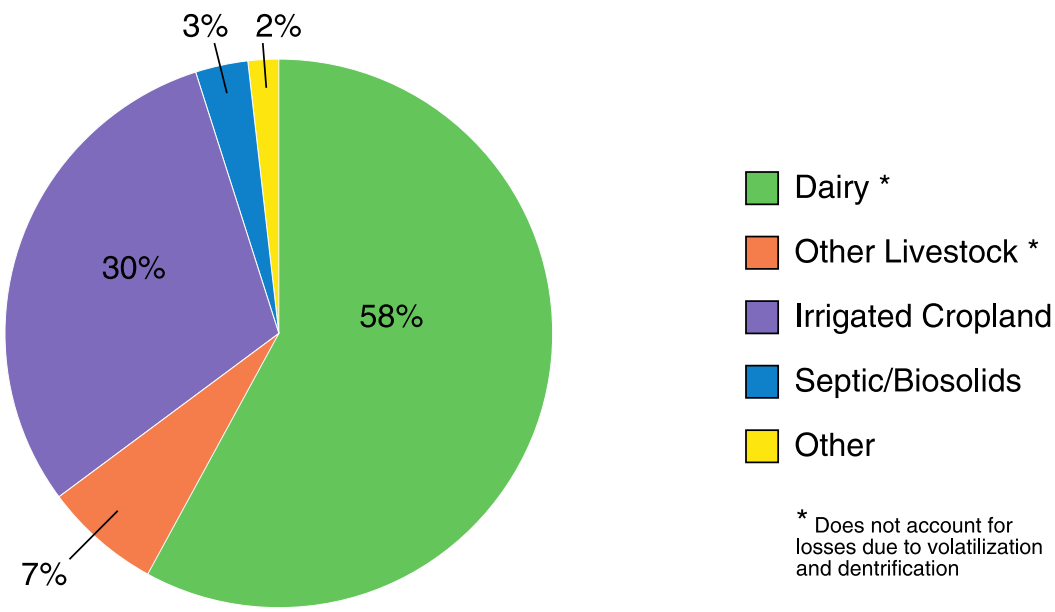

Figure 6: Nitrogen Generated by Major Sources in Yakima County

Turning to northwestern Whatcom County, Ecology has documented ground water contamination in the Sumas-Blaine Aquifer in northwestern Whatcom County in the *Sumas-Blaine Aquifer Nitrate Contamination Summary*.[12] That study documents widespread nitrate contamination in ground water of the Sumas-Blaine aquifer for over forty years. *Id.* at 7. The study pointed out that Washington State, and the Sumas-Blaine aquifer in particular, is

[12] *See* Barbara Carey, Washington State Department of Ecology, *Sumas-Blaine Aquifer Nitrate Contamination Summary* (2012), *available at* https://fortress.wa.gov/ecy/publications/publications/1203026.pdf.

especially susceptible to ground water contamination by the keeping of animals due to several factors. *Id.*

A study recently released in March 2014 quantified the mass of nitrogen added to [a manured grass field overlying the Sumas-Blaine Aquifer] in the form of manure, inorganic fertilizer, and irrigation water; the mass of nitrogen removed in the crop; the mass of nitrate in the soil found during the post-harvest period; and the nitrate contamination of groundwater beneath the field.[13] The study noted that the shallow groundwater depth "enabled rapid responses to nitrate transport, especially during the high rainfall period." *Id*. Given the extent of the contamination detected, the study concluded that "[d]irect monitoring of water quality at the water table was the only accurate and reliable method for tracking efforts of manure management on groundwater nitrate." *Id*.

Shallow water tables, like those found on the West of the Cascades, result in greater contamination levels because compounds do not need to travel through the soil as far to reach the water table. This is especially true for water tables that can reach the bottom of manure lagoons during the seasonal high level of the groundwater table, resulting in direct leakage of nitrates from lagoon to the groundwater.[14] Across most of the Sumas-Blaine aquifer, the depth to ground water is less than ten feet, making it highly susceptible to contamination from agricultural activities. Carey, *supra* note 12.

In addition to a shallow water table, the seasonal high rainfall of the region further exacerbates the rate at which nitrates contaminate the groundwater.[15] When manure is

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[13] Barbara Carey, Ecology Nitrogen Dynamics at a Manured Grass Field Overlying the Sumas-Blaine Aquifer in Whatcom County, Ecology Publication No. 14-03-001 (March 2014).

[14] Melanie Kimsey, Wash. State Dep't of Ecology, *Constr. of Dairy Lagoons Below the Seasonal High Ground Water Table*, 4 (2002).

[15] J.M. Ham et al., Kansas State University, *Animal Waste Lagoon Water Quality Study*, 2.4 (1999), *available at* http://www.ksre.ksu.edu/bookstore/pubs/LAGOON.pdf.

applied to fields the nitrates need to be utilized by the crops or the nitrates will easily be washed away by rain into waterways or into the soil through over-application eventually reaching the water table.[16] Washington has a high level of precipitation during the non-growing season, carrying nitrates left over in the soil to the water table. *Id.* at 30.

Nitrate contamination is so severe in the Sumas-Blaine aquifer that several public supply wells near the City of Lynden were shut-down, leaving 1,200 people without a potable water supply. *Id.* at 9. In 2008, a study reported that sampled wells displayed an increasing trend in groundwater nitrate levels, while another study reported that seventy-one percent of thirty-five private wells sampled were over the drinking water MCL.[17] Whatcom County has over 40,000 cows [an accurate number of cows is not possible given the information withheld pursuant to WAC 16.06.210(29)], a shallow water table, and heavy rainfall during the non-growing season, resulting in some of the highest nitrate ground water levels in the state. *See generally* Barbara Carey, Washington State.[18]

Washington's Sumas-Blaine aquifer illustrates the dangers of nitrate leaching into ground water. Above the Sumas-Blaine aquifer, soil nitrate concentrations at one site ranged up to 240 percent higher than the "very high" criterion suggested by one study. Carey, *supra*, at 69. The condition of the Sumas-Blaine aquifer exemplifies the overall effect of these elevated nitrate concentrations on the quality of the public's drinking water. In forty years of research, the Department of Ecology found that twenty-nine percent of the 515 sampled wells exceeded

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[16] Barbara M. Carey, Wash. State Dep't of Ecology, *Effects of Land Application of Manure on at Two Dairies over the Sumas-Blaine Surficial Aquifer*, viii (2002).

[17] Melanie Redding, Washington State Department of Ecology, *Nitrate Trends in the Central Sumas-Blaine Surficial Aquifer* 23 (2008), *available at* https://fortress.wa.gov/ecy/publications/publications/0803018.pdf at 43.

[18] Department of Ecology, *Sumas-Blaine Aquifer Nitrate Contamination Summary* (2012).

acceptable nitrate concentrations for drinking water in the Sumas-Blaine aquifer—the primary drinking water source for between 18,000 and 27,000 people.[19]

The contamination of drinking water and ground water is sufficiently pervasive to have reached the attention of state courts. The Eastern District of Washington, after hearing evidence regarding contamination of ground water due to the keeping of animals at a CAFO found that "excessively high levels of nitrate and phosphorus are consistent with over-applications of manure" at the Nelson Faria Dairy in Royal City, Washington. *CARE v. Nelson Faria Dairy*, Case No. CV-04-3060-LRS (Memorandum of Decision) (December 30, 2011) (Exhibit A). The court further concluded that "Faria's manure management practices are the predominant source of the nitrate contamination found in the monitoring wells and, correspondingly, local groundwater. These practices include consistent over-application of manure to fields located adjacent to, and nearby, the Dairy." *Id*.

A conservative estimate of the number of cows in Whatcom County in 2010 is 40,834. Carey, *supra*, at 19. Again, it is impossible to verify this number given the data that is withheld pursuant to WAC 16.06.210(29). Between ten and twelve million pounds of manure-derived nitrate is applied to land over the Sumas-Blaine aquifer annually. *Id*. In 1990, a study sampled twenty-seven wells of the Sumas-Blaine aquifer and reported that twenty-six percent of the wells exceeded the nitrate MCL of ten mg/l-N. *Id.* A similar study in 2012 found that thirty-six percent of shallow wells exceeded the MCL and the highest nitrate concentration in a domestic drinking well was seventy-three mg/l-N. Carey, *supra*, at 7.

Similarly, in the Lower Yakima Valley in Washington State, about twenty percent of sixty-seven wells sampled in 2012 exceeded the MCL of ten mg/l-N.[20] Dairy CAFOs, like those

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[19] Wash. State Dep't of Ecology, *Report Summarizes 30 Years of Nitrate Studies in the Sumas-Blaine Aquifer* (2012), *available at* https://fortress.wa.gov/ecy/publications/publications/1203026.pdf.

investigated by EPA in the Lower Yakima Valley study, are a known contributor of nitrate contamination to ground water through two mainstream agricultural processes: land application of manure and dairy lagoons.

a. LAND APPLICATION OF ANIMAL MANURE

A major source of anthropogenic nitrate contamination to ground water is cropland over-application of manure as fertilizer. Over-application is due to using manure as fertilizer above the suggested agronomic rate for uptake of manure's component compounds by the soil, or applying to frozen or saturated soils when uptake is impeded. Many studies document that over-application of manure, which results in excessive nitrate concentrations in the soil, is a widespread problem with deleterious impacts on surface and ground water quality and its effects on ground water quality.[21]

Elevated nitrate concentrations in the soil can harm ground water quality, "because organic nitrogen from manure that accumulates in soil gradually mineralizes to nitrate. If not biologically taken up, nitrate from mineralized organic nitrogen is available for leaching" into ground water. Carey, *id.*, at viii. Excessive manure application has been found to adversely affect ground water quality and result in nitrate concentrations above the MCL,[22] and the data

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[20] U.S. EPA Region ten, *Relation Between Nitrate in Water Wells and Potential Sources in the Lower Yakima Valley*, ES-3 (2013).

[21] *See generally* Carey, *supra* at viii; Denis Erickson & Wym Matthews, Wash. State Dep't of Ecology, *Effects of Land Application of Dairy Manure and Wastewater on Groundwater Quality* (2002), *available at* https://fortress.wa.gov/ecy/publications/publications/0203002.pdf; Barbara M. Carey, Wash. State Dep't of Ecology, *Effects of Land Application of Manure on Groundwater at Two Dairies over the Sumas-Blaine Surficial Aquifer* (2002), *available at* https://fortress.wa.gov/ecy/publications/publications/0203002.pdf; Redding, *supra*; Jorge A. Delgado et al., *Advances in Nitrogen Mgmt. for Water Quality*, 379 (2010).

[22] Erickson & Matthews, *id.*, at 42; Carey, *supra*, at vii-viii (one site exhibited average nitrate concentrations up to double the drinking water standard); Jorge A. Delgado et al., *Advances in*

demonstrates that the application methods of several CAFOs in Washington exceed the suggested agronomic rate.[23] A California study found that cropland was the largest nitrate source at ninety-seven percent of all nitrate leached to ground water, with manure application being one of the nitrogen sources on cropland.[24]

In spite of the evidence of nitrate contamination of groundwater and drinking water due to dairies, AFOs and CAFOs cited above, Washington needs to take a hard look to ascertain whether all AFOs are applying manure at agronomic rates. Ecology has attempted to get such information by requiring soil sampling as part of the WA CAFO General NPDES permit that was issued in 2006.[25] However, as previously mentioned, only approximately 13 of over 1200 CAFOs in the state are covered by the 2006 CAFO General Permit, a result that violates the Clean Water Act. Ecology's efforts are a tiny fraction of what is needed. Even if the Department of Agriculture gathered this much-needed information as to whether manure is being applied agronomic rates at all dairies, AFOs and CAFOs, that information would be

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Nitrogen Management for Water Quality (2010); Carey, Washington State Department of Ecology, *Sumas-Blaine Aquifer Nitrate Contamination Summary* (2012); Redding, *supra*, 23; Jose Delgado & W.C. Bausch, *Potential use of precision conservation techniques to reduce nitrate leaching in irrigated crops,* (2005).

[23] Erickson & Matthews, *supra* at 46; Carey, *supra* at vii-viii (one site's application method was double the suggested agronomic rate); Jorge A. Delgado et al., Advances in Nitrogen Management for Water Quality 379 (2010); Barbara Carey, Washington State Department of Ecology, *Sumas-Blaine Aquifer Nitrate Contamination Summary* (2012).

[24] Thomas Harter et al., State Water Resources Control Board, *Addressing Nitrate in California's Drinking Water* 17 (2012).

[25] Commenters do not believe that soil sampling is the best method to obtain information on agronomic rates. Only groundwater monitoring can give a full and accurate picture as to whether nitrates are leaching through the soil and into the groundwater. Groundwater monitoring must be a part of any definition of "best agricultural practices" in regards to large CAFOs.

essentially exempt from public disclosure based upon WAC 16.16.210(29), even though such information is clearly "meaningful information to the public."

b. MANURE STORAGE LAGOONS

Studies demonstrate that manure storage lagoons contaminate ground water due to seepage of nitrates.[26] Indeed, any engineer will tell you that manure lagoons are designed to leak. It is a principle of geology known as Darcy's Law.[27] Several factors contribute to an increased rate of nitrate seepage, including the construction method of the basin liner (if any) and the vertical separation distance to the water table (if known).[28]

Lagoon basins to store manure can be constructed as earthen-lined basins, concrete-lined basins, or unlined basins.[29] Here in Washington, it is believed that only one CAFO in the state has a synthetically-lined lagoon, but this information on dairies, AFOs and CAFOs not covered by a discharge permit would not be subject to disclosure even though this information is specifically relevant to human health and the environment. Studies show that regardless of

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[26] Thomas Harter et al., *Addressing Nitrate in California's Drinking Water*, 17 (2012), *available at* http://groundwaternitrate.ucdavis.edu/files/138956.pdf; *see also* J.M. Ham et al., Kansas State University, *Animal Waste Lagoon Water Quality Study*, 2.4 (1999), *available at* http://www.ksre.ksu.edu/bookstore/pubs/LAGOON.pdf.

[27] Darcy's law is "a law in geology describing the rate at which a fluid flows through a permeable medium. Darcy's law states that this rate is directly proportional to the drop in vertical elevation between two places in the medium and indirectly proportional to the distance between them. The law is used to describe the flow of water from one part of an aquifer to another and the flow of petroleum through sandstone and gravel." *See* http://dictionary.reference.com/browse/darcy%27s%20law (last visited March 26, 2014).

[28] *See, e.g.,* Minnesota Pollution Control Agency, Effects of Liquid Manure Storage Systems on Ground Water Quality 93-6 (2001), *available at* http://www.pca.state.mn.us/index.php/view-document.html?gid=6336.

[29] Robbin Marks, Cesspools of Shame: How Factory Farm Lagoons and Sprayfields Threaten Environmental and Public Health, 33 (2001), *available at* http://www.nrdc.org/water/pollution/cesspools/cesspools.pdf.

the materials used for lining basins, if you monitor the ground water, you will find evidence of ground water contamination down-gradient from manure storage areas.[30] Some types of liners are less effective than others; unlined basins have been shown to impact ground water monitoring wells.[31] Researchers generally agree that leakage rates decrease after lagoons first receive waste, but these lower leakage rates still pose a threat to ground water.[32] For example, one study concluded that, even after a lagoon undergoes optimum sealing, seepage of 0.1 mm/day is likely. *Id*.

In addition to the construction method of basin liners, the vertical separation distance of the lagoon to the water table is a crucial factor in determining the lagoon's contribution to contaminating ground water.[33] In areas with a seasonally high ground water table the contamination of ground water will occur more readily. *Id.* "A lagoon constructed below the seasonal high ground water table is essentially a direct discharge to ground water."[34]

One California study, measuring nitrate contamination of groundwater in two central California watersheds, established a benchmark to distinguish between low and high intensity

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[30] *See* Minnesota Pollution Control Agency, *Effects of Liquid Manure Storage Systems on Ground Water Quality*, 1 (2001), *available at* http://www.pca.state.mn.us/index.php/view-document.html?gid=6336.

[31] *The impact of dairy farms on groundwater quality in Israel's Coastal Aquifer*, Ben-Gurion University of the Negev, Israel, (2010).

[32] Denis Erickson, Wash. State Dep't of Ecology, *Edaleen Dairy Lagoon Ground Water Quality Assessment* 1 (1991), *available at* https://fortress.wa.gov/ecy/publications/publications/91e11.pdf.

[33] Jay M. Ham, *Seepage Rates and Nitrogen Losses from Animal Waste Lagoons: Potential Impacts on Groundwater Quality*, 1 (2000).

[34] Melanie Kimsey, *Washington State Dep't of Ecology, Construction of Dairy Lagoons Below the Seasonal High Ground Water Table*, 4 (2002).

nitrate leaching of thirty-five kg N/ha/yr.[35] "Total nitrate loading to groundwater above this benchmark indicates a high potential for regional groundwater degradation." *Id*. This study found that, although animal corrals and manure storage lagoons combined accounted for only 0.8 percent of total nitrate leaching in the study area, the average intensity of nitrate loading to groundwater from these sources was 183 Kg N/ha/yr. *Id*. at 18. Cumulatively, not accounting for high water tables, the rates of seepage can exceed 250,000 lbs of nitrate losses to ground water from one CAFO lagoon over the course of a twenty-year operation. Ham et al., *supra*, 2.4.

c. Dairy, AFOs & CAFOs Cause Surface Water Pollution

The extent of surface water discharges from dairies, AFOs and CAFOs is also well documented. Just last week, a Skagit dairy farm (Beaver Marsh Dairy) was fined for applying liquid manure to a field which caused a discharge to a water of the state which raised fecal coliform levels.[36] One only need to view public records at Ecology or the case law to see the scope of the problem in regards to surface water discharges from dairies, AFOs and CAFOs.[37] Amazingly, even in the face of clear evidence of facilities discharging to waters of the state, Ecology has not required coverage under the CAFO General Permit or under individual discharge permits.

The discharges to surface water are also reflected in the rampant surface water contamination due to nutrient pollution. Ocean acidification is a game changer here in Washington State. Acidification of our marine waters has the potential to alter the marine food

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[35] Thomas Harter et al., State Water Resources Control Board, *Addressing Nitrate in California's Drinking Water* 17 (2012) at 16-17.

[36] "Skagit County Dairy Fined $6000 for Manure Pollution in Beaver Marsh," *available at* http://agr.wa.gov/News/2014/14-07.aspx (last visited March 27, 2014).

[37] *See, e.g.,* "AFO/CAFO Discharge Information: 2008-2010," obtained from WA Department of Ecology CAFO files in response to public records request (listing examples of discharges from CAFOs by Regional Office) (Exhibit D); *CARE v. Bosma Dairy* (9th Cir. 2002).

web with unimaginable environmental and economic consequences for present and future generations of Washingtonians. In November 2012, the Washington State Panel on Ocean Acidification issued its Summary Report, *Ocean Acidification From Knowledge to Action*. Appendix 8, *Washington State's Legal and Policy Options for Combating Ocean Acidification in State Waters*, and identified the nutrient pollution discharged from operations that keep animals as a major land-based contributing factor to ocean acidification:

> It is likely that some coastal pollutants, such as nutrient runoff, exacerbate the effects of atmospheric-CO2-driven acidification in nearshore waters, magnifying impacts on shellfish and other marine organisms. Because these pollutants originate within Washington, the State generally has the authority to curtail them, offering a means of partially alleviating the effects of acidification in State waters.

Appendix 8 p. 5. The report specified:

> The sources of nutrient runoff include wastewater treatment, septic systems, residential fertilizer, stormwater, *dairy operations*, crop agriculture, *livestock*, and increased terrestrial erosion.

Appendix 8 p. 9 (emphasis added). The report specifically identified dairies, AFOs and CAFOs, and the manure that they produce as significant a significant contributing factor to the state's ocean acidification crisis:

> Dairy operations and other sites of high manure concentrations are important sources of nutrient inputs into Puget Sound and other Washington State waters. There are approximately 517 dairy farms in the State, producing many thousand tons of manure annually. This manure is high in nitrogen and phosphate compounds, and can contribute to eutrophication if these nutrients are released into regional watersheds and into the coastal ocean to which those watersheds drain. This eutrophication, in turn, can cause algal blooms and contribute to coastal ocean acidification as described above.

Appendix 8 p. 23. Because of the challenges associated with drawing down greenhouse gas ("GHG") emissions in the urgent timeframe needed to address the rapid pace ocean

acidification, it is imperative that we utilize all existing legal authority to address the land-based contributing factors if we are to have any hope of maintaining the marine food web for present and future generations. Land-based activities, such as the discharge of manure from industrial animal operations are clearly within the scope of existing state regulatory control. But we cannot do that without the requisite data, including how much pollution is generated by dairies, AFOs and CAFOs.

The closure of numerous shellfish beds in the Puget Sound region have been directly linked to nutrient pollution caused by dairies, AFOs and CAFOs, especially dairy cows.[38] This is a serious public health threat. The shellfish industry should not suffer do to the lack of enforcement of regulations or the withholding of vital information. According to the Ecology, the sources of bacteria pollution to Samish Bay "include surface flow from areas where livestock or manure application is occurring during storm events, malfunctioning on-site septic systems, waterfowl and wildlife, stormwater runoff, pets, non-commercial farm animals, and recreational users."[39] In 2011, the Department of Health downgraded 4,037 acres of intertidal shellfish growing areas in Samish Bay to "conditionally approved" for commercial shellfish harvest.[40] In 2012, Samish Bay commercial shellfish beds were closed to harvest for a total of 81 days. As of December 5th, Samish Bay commercial shellfish beds have been closed for a total

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[38] The average dairy cow produces 14.3 gallons of wet raw manure per day. *See* CAFO Amended Fact Sheet, June 21, 2006.

[39] Samish Bay Watershed Fecal Coliform Bacteria Total Maximum Daily Load Volume 1: Water Quality Findings. Publication 08-03-029, November 2008.

[40] 201 Annual Report: Commercial and Recreational Shellfish Areas in Washington State, Washington State Department of Health, Office of Shellfish and Water Protection, September 2012).

of 59 days so far in 2013.[41] "[F]ecal coliform pollution has resulted in closures for recreational shellfish harvest and swimming at Bay View State Park [in Skagit County] over the years. The park's beach is permanently closed to shellfish harvest until pollution sources are identified, correct and water quality improves significantly . . . ."[42]

In 2007 Food and Water Watch estimated that Skagit County had 7,589 dairy cows capable of producing 39.6 million gallons of manure per year, excluding other heifers and other types of livestock.[43] It is impossible to verify this estimate due to the existing language in WAC 16.06.210(29). A study found that this manure ends up in the ground and surface waters of this region, as reflected by the "spike in fecal coliform found in the Samish watershed in 2008."[44] In sum, the data that has been collected or estimated, and the resulting shellfish bed closures, demonstrates that dairies, AFOs and CAFOs are being operated in a way that endangers human health and the environment. The people have a right to information about this to protect their rights to gather, consume and enjoy the shellfish resources in thi state.

In addition to the obvious link between surface water pollution and degradation of salmon habitat, there is also a tremendous amount of scientific data confirming that the protection of the state's ground water resources is critical to restore and protect salmon. "Agricultural non-point source water pollutants such as sediment, pesticides and nutrients have been identified as contributing to the environmental distress of salmon runs in the Pacific

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[41] Washington State Department of Health, Office of Shellfish and Water Protection, Personal Communication with Scott Berbells 360-236-3324).

[42] *See infra* note 46.

[43] Factory Farm Map, a project of Food & Water Watch, *available at* http://www.factoryfarmmap.org/ (last visited December 6, 2013).

[44] "Mixed Results for Skagit Water Quality," Skagit Valley Herald (Nov. 23, 2013), *available at* http://www.goskagit.com/all_access/mixed-results-for-skagit-water-quality/article_d011e6bf-223a-5c7a-a341-b496d8a258ee.html (last visited Dec. 6, 2013).

Northwest."[45] Therefore, the Board's regulatory decisions, or lack thereof, regarding the keeping of animals has a strong potential to affect habitat for one of our state's most critical natural resources: salmon. This is due to the hydrological connection between the surface and ground waters in this state. For example:

> These circumstances make groundwater a crucial component of river habitats. Groundwater can influence the distribution, reproductive success, biomass and productivity, behaviour and movements of fishes, and is especially important in winter and summer. Winter flows are minimal and are affected by ice. In winter, the importance of groundwater increases northwards.
>
> * * *
>
> Almost all of the water flowing in rivers and streams is derived from groundwater, very little is direct fallout or overland run-off. For this reason, the quantity and quality of water is determined by the sources and characteristics of groundwater, exchanges between river water and groundwater along the stream, and water conditioning in the exchange zones of the stream bed and at the stream surface.[46]

The protection of salmon from nutrient pollution is a public health issue.

> Large quantities of finfish and shellfish are caught each year, both recreationally and commercially, and many residents eat seafood harvested from our waters. In addition, tribal populations enjoy treaty fishing rights, and

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[45] Whittaker, G. 2005. Application of SWAT in the evaluation of salmon habitat remediation policy. *Hydrological Processes*, 19, 839-848.

[46] Power, G., Brown, R.S., and Imhof, J.G. 1999. Groundwater and Fish insights from northern North America. *Hydrological Processes,* 13, 401-422 (In particular, nutrients and sediment from agricultural activities are seen as damaging to salmon and salmon habitat (Pacific Fisheries Management Council, 1999). Nitrogen fertilizer entering streams is considered to present a hazard to salmon. In particular, nutrient loading is thought to be responsible for 'increased primary and secondary production, possible oxygen depletion during extreme algal blooms, lower survival and productivity, increased eutrophication rate of standing waters, certain nutrients (e.g., non-ionized ammonia, some metals) possibly toxic to eggs and juveniles at high concentrations' (Pacific Fisheries Management Council, 1999)").

> harvesting and eating seafood plays a significant role in their cultures. Finfish and shellfish are important parts of a healthy diet.[47]

The present and future generations of this state should not only be able to eat salmon and shellfish from Washington waters, but they should be able to do so without concern that the fish consumed is contaminated with pollutants caused by dairies, AFOs and CAFOs.

One of the most urgent ecological crises that Puget Sound faces today is the health of its Endangered Southern Resident killer whales. These three pods, J-Pod, K-Pod and L-Pod, were decimated by the accumulation of marine toxins and the depletion of prey resources primarily caused by the degradation of salmon spawning and nearshore habitats, the nurseries of the Salish Sea.

As discussed above, insufficiently regulated agricultural practices are degrading water quality throughout the state. Even small changes in stream temperature and water quality can impact salmon populations. Poor stream quality can also impact marine water quality by introducing nutrients that can result in algal blooms, chemical and pathogen pollution, and lowering oxygen levels in areas where these streams empty into the habitat of our resident killer whales. Since a healthy killer whale population depends upon strong salmon runs, it is imperative that the Board of Health retains and implements its existing authority to ensure that animals are not kept in a manner that further degrades the remaining salmon habitat that feeds the Salish Sea.

Therefore, the science is clear that the amount of waste generated by dairies, AFOs and CAFOs is so great that how it is managed has the potential to significantly affect human health and the environment. The only way the public can take steps to protect itself from these operations, and to ensure that the government is doing what it is supposed to do in regards to

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[47] Washington Department of Ecology, *Fish Consumption Rates, Technical Support Document: A Review of Data and Information about Fish Consumption in Washington* (January 2013).

protecting human health and the environment, is if it has the information needed to do so. That includes meaningful information regarding: (1) Number of animals; (2) volume of livestock nutrients generated; (3) number of acres covered by the plan or used for land application of livestock nutrients; (4) livestock nutrients transferred to other persons; and (5) crop yields. As seen in the studies discussed above, only with reliable information on each of these categories of information can a citizen ascertain the amount of pollution being discharged by the facility into our commonly-shared air and water resources. Because Washington citizens have inherent rights to ensure their public health and a clean environment, they are entitled to information that affects those rights.

5. Dairies, AFOs and CAFOs Affect the Public Interest By Threatening Public Health

The threat to public health by dairies, AFOs and CAFOs is well-documented. In 2010 the Environmental Protection Agency (EPA) conducted well water testing in the lower Yakima Valley in order to determine nitrate levels in drinking water and also to better characterize contamination of the ground water. EPA analyzed for almost 200 separate chemicals. In dairy supply wells that ranged in depth from 200 to 430 feet the EPA found the herbicide bentazone; the veterinary pharmaceuticals monensin, tetracycline and virginiamycin; and the synthetic and natural hormones 17 beta estradiol, 17 alpha estradiol, 17 beta trenbolone, alpha zearalenol, beta zearalenol and androstenedione and testosterone (EPA, 2013A). Some of these chemicals are known hormone disruptors and should not be tolerated at any level in our drinking water.

The drinking water standard for nitrate is ten mg/l-nitrogen (N) for Washington State.[48] There is abundant evidence that nitrate levels exceeding the ten mg/l-N standard can pose a

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[48] Melanie Redding, Washington State Department of Ecology, *Nitrate Trends in the Central Sumas-Blaine Surficial Aquifer* 23 (2008), *available at* https://fortress.wa.gov/ecy/publications/publications/0803018.pdf.

public health threat.[49] Methemoglobinemia in infants (blue baby syndrome), increased risks for pregnant women and individuals with digestive problems, and increased rates of adult cancer are but a few of the health issues correlated to elevated nitrates in drinking water.[50] Because ground water accounts for the drinking water of more than thirty-three percent of the United States, contamination is not an issue of isolated concern.[51] This is especially true in Washington state where there is a high level of hydrologic connectivity between surface and ground water.[52]

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[49] *Id.; see also* Jeff Feaga et al., Oregon State University Agricultural Experiment Section, *Nitrates and Groundwater: Why Should We Be Concerned With Our Current Fertilizer Practices?* 1 (2004); Karen R. Burow et al., *Nitrate in Groundwater of the United States, 1991-2003*, 44 Environmental Science & Technology 4988, 4988 (2010) ("Contamination of ground water by nitrate is of concern because elevated concentrations can affect human health."); Bernard T. Nolan & Kerie J. Hitt, *Vulnerability of Shallow Groundwater and Drinking-Water Wells to Nitrate in the United States*, 40 Environmental Science & Technology 7834, 7834 (2006) ("High nitrate concentration in ground water is a human health concern.").

[50] Redding, *supra*, at 23.

[51] Karen R. Burow et al., *Nitrate in Groundwater of the United States, 1991-2003*, 44 Environmental Science & Technology 4988, 4988 (2010) at 4988.

[52] Robert M. Hirsch, Chief Hydrologist, from the forward to *Ground Water and Surface Water: A Single Resource*, U.S. Geological Survey Circular 1139 (1998) ("Nearly all surface-water features (streams, lakes, reservoirs, wetlands, and estuaries) interact with groundwater . . . . [P]ollution of surface water can cause degradation of ground-water quality and, conversely, pollution of ground water can degrade surface water."); Vaccaro, J., *River Aquifer Exchanges in the Yakima River Basin, Washington*, U.S. Geological Survey Scientific Observations Report 2011-5026 (2011) ("Most of the year, streamflow in the Yakima River basin is largely baseflow or groundwater that has discharged to the stream channel; therefore, the quality and availability of surface water are largely influenced by groundwater. Perennial streams are supported by groundwater and constitute a groundwater-dependent ecosystem (GDE) (Hatton and Evans, 1998; Eamus and Froend, 2006). Riparian habitat, and algal, invertebrate, and fish communities therefore are, to some extent, dependent on groundwater discharge to perennial streams.").

A general survey of publically available scientific studies illustrates that instances of high levels of nitrate can be found in several counties throughout Washington state. A study conducted in Grant, Adams, and Franklin Counties found that twenty-three percent of overall samples exceeded the ten mg/l-N level and an additional thirty-seven percent of the samples had nitrate concentrations between three and ten milligrams per liter, constituting a large group of wells with elevated concentrations of nitrate.[53] In Franklin County, thirty percent of the samples exceeded the maximum contaminant level (MCL), while approximately twenty percent of the samples in both Grant and Adams Counties exceeded the MCL. *Id*.

One phase of the EPA study in the Lower Yakima Valley found nitrate levels above the safe drinking water standard of 10 mg/L in 20% of domestic wells. A later sampling found one monitoring well down gradient from a dairy with nitrate levels of 190 mg/L (EPA, 2013 B, p.7). The impact for people who live in the Yakima Valley, and elsewhere in the state, is uncertainty related to risks. Concerned families incur added costs to drill new wells, or purchase well water testing, water treatments and bottled water. A poverty level family of four can easily spend 5% of their annual income just for safe drinking water. But that is not the moral or just result. Rather, by making accurate data available regarding the amount of nutrients generated by unpermitted dairies, AFOs and CAFOs throughout the state, those individuals and entities who produce the nutrients, and benefit economically therefrom, can take steps to ensure that the nutrients are being produced in a manner that does not threaten public health and the environment.

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[53] Sarah J. Ryker & Lonna M. Frans, Water-Resources Investigations Report 99-4288, *Summary of Nitrate Concentrations in Ground Water of Adams, Franklin, and Grant Counties, Washington, Fall 1998—A Baseline for Future Trend Analysis, U.S. Department of the Interior* 1 (2000), *available at* http://pubs.usgs.gov/wri/1999/4288/report.pdf.

In February 2013, the Washington Department of Health investigated a cluster of neural tube defects in Central Washington from 2010-2013.[54] The study recommended "monitoring private well nitrate concentrations because of their potential association with birth defects and other adverse health outcomes." *Id.* In 2009, a Washington Department of Health study found that "exposure to nitrate from drinking water significantly and substantially increases the risk of an infant having physiologically elevated levels of methemoglobin."[55] The National Association of Local Boards of Health stated that "[t]he most pressing public health issue associated with CAFOs stems from the amount of manure they produce. CAFO manure contains a variety of potential contaminants."[56] The study noted that "[r]egular testing of household water wells for total and fecal coliform bacteria is a crucial element in monitoring groundwater quality, and can be the first step in discovering contamination issues related to CAFO discharge." *Id*. But the onus cannot be only on the public to protect themselves from CAFO pollution. By withholding critical information pursuant to WAC 16.06.210(29), the Department of Agriculture is making it virtually impossible for members of the public to ascertain whether their health is at risk due to CAFO pollution. That result is unjust and illegal.

6. Dairies, AFOs and CAFOs Affect the Public Interest By Contributing to Climate Change

Dairies, AFOs and CAFOs are major contributors of greenhouse gas emissions ("GHG") in the state of Washington. "Agricultural activities such as manure management, fertilizer use, and livestock (enteric fermentation) result in methane and nitrous oxide emissions that account

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[54] Washington Department of Health, Notes from the Field: Investigation of a Cluster of Neural Tube Defects – Central Washington, 2010-2013.

[55] James VanDerslice, WA State Department of Health, Well Water Quality and Infant Health Study (January 14, 2009).

[56] Carrie Hribar, National Association of Local Boards of Health, Understanding Concentrated Animal Feeding Operations and Their Impact on Communities (2010) at 2.

for 6% of State GHG emissions in 2005."[57] Worldwide, the livestock sector generates more GHG emissions as measured in CO2 equivalent (18%) than the transportation sector.[58] Livestock generates 65% of human-related nitrous oxide which has 296 times the global warming potential of CO2, accounts for 37% of all human-induced methane (23 times as warming as CO2 and is responsible for 64% of ammonia emissions: devastating health effects. *Id.* "Global greenhouse gas emissions from the agricultural sector totaled 4.69 billion tons of carbon dioxide (CO2) equivalent in 2010 (the most recent year for which data are available), an increase of 13 percent over 1990 emissions. By comparison, global CO2 emissions from transport totaled 6.76 billion tons that year, and emissions from electricity and heat production reached 12.48 billion tons, according to Worldwatch Institute's Vital Signs Online service (www.worldwatch.org)."[59] The over-application of manure has been identified as a major contributing factor to increased GHG emissions:

> Manure that is deposited and left on pastures contributes to global nitrous oxide emissions because of its high nitrogen content. When more nitrogen is added to soil than is needed, soil bacteria convert the extra nitrogen into nitrous oxide and emit it into the atmosphere—a process called nitrification. Emissions from manure on pasture were highest in Asia, Africa, and South

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[57] WA Department of Community, Trade & Economic Development, Washington State Greenhouse Gas Inventory and Reference Case Projections, 1990-2010 (December 2007), *available at* http://www.ecy.wa.gov/climatechange/docs/WA_GHGInventoryReferenceCaseProjections_1990-2020.pdf (last visited March 31, 2014).

[58] Livestock's Long Shadow – Environmental Issues and Options, United Nations Food & Agriculture Organization (Nov. 29, 2006).

[59] Worldwatch Institute, "Agriculture and Livestock Remain Major Sources of Greenhouse Gas Emissions," available at http://www.worldwatch.org/agriculture-and-livestock-remain-major-sources-greenhouse-gas-emissions-1 (last visited March 31, 2014).

America, accounting for a combined 81 percent of global emissions from this source.[60]

In Washington, "[t]he manure management category [of emissions], which shows the highest rate of growth relative to the other categories, accounted for 11% [] of total agricultural emissions in 1990 and Is estimated to account for about 25% [] of total agricultural emissions in 2020."[61] The science is clear that livestock population is a critical component of any emissions calculation for the agricultural sector. *Id.* "For dairy cattle and heifers, the proportion of manure managed in systems that yield higher GHG emissions (e.g. anaerobic lagoons and liquid slurry) than other systems (e.g., pasture) increased from 68% for dairy cattle and 71% for dairy heifers in 1990, to 76% for dairy cattle and 77% for dairy heifers for 1997 through 2002. *Id*. The GHG emissions calculations done in Washington for the agricultural sector explicitly recognize the need for more precise data because "[e]missions from enteric fermentation and manure management are dependent on the estimates of animal populations and the various factors used to estimate emissions for each animal type and manure management system (i.e., emission factors which are derived from several variables including manure production levels, volatile solids content, and CH_4 formation potential)." *Id*. at F-6. Given the fact that climate change is the most pressing challenge, work to reduce GHG emissions from the agricultural sector should not be hamstrung by reliance on "estimates" instead of actual data. It is imperative that DOA amend WAC 16.06.210(29) so that meaningful information on dairies, AFOs and CAFOs is publically available.

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[60] *Id.*

[61] WA Department of Community, Trade & Economic Development, Washington State Greenhouse Gas Inventory and Reference Case Projections, 1990-2010 (December 2007), *available at* http://www.ecy.wa.gov/climatechange/docs/WA_GHGInventoryReferenceCaseProjections_1990-2020.pdf (last visited March 31, 2014) at F-4.

b. The Five Categories of Information Are Crucial Pieces of Information That Affect the Public Interest

The existing WAC 16.06.210(29) effectively creates an exemption for five categories of information, all of which are crucial pieces of information to ascertain (1) the extent of pollution coming from these operations; and (2) whether the government is fulfilling its statutory obligations to protect public health and the environment. The five categories of information are (1) Number of animals; (2) volume of livestock nutrients generated; (3) number of acres covered by the plan or used for land application of livestock nutrients; (4) livestock nutrients transferred to other persons; and (5) crop yields. The Legislature has recognized the significance of these categories of information by directing that "meaningful information" about each category be provided to the public. RCW 42.56.610; 90.64.190.

That makes practical sense. If you do not know how many animals a dairy, AFO or CAFO has, it is impossible to determine how much pollution is generated at the facility. The average dairy cow produces 14.3 gallons of wet raw manure per day. *See* Ecology CAFO Amended Fact Sheet, June 21, 2006. For mature dairy cattle, WAC 16.06.210(29) requires ranges that are so wide as to be meaningless. For comparison purposes, below is a table of the existing ranges and how much wet raw manure is produced at each end of the range on a daily and annual basis:

Dairy Cows' Manure Produced (Average[62] by Category)		
Number of Mature Dairy Cattle[63]	Manure Produced Daily (gallons)	Manure Produced Annually (gallons)

[62] CAFO General Permit Amended Fact Sheet (June 21, 2006), *available at* http://www.ecy.wa.gov/programs/wq/permits/cafo/cafo_final_fs.pdf, at 11 (last visited March 31, 2014).
[63] WAC 16-06-210(29).

1	37	14.3	529.1	5,219.5	193,121.5
38	199	543.4	2,845.7	198,341	1,038,680.5
200	699	2,860	9,995.7	1,043,900	3,648,430.5
700	1,699	10,010	24,295.7	3,653,650	8,867,930.5
1,700	2,699	24,310	38,595.7	8,873,150	14,087,430.5
2,700	3,699	38,610	52,895.7	14,092,650	19,306,930.5
3,700	4,699	52,910	67,195.7	19,312,150	24,526,430.5
4,700	5,699	67,210	81,495.7	24,531,650	29,745,930.5
5,700	6,839	81,510	97,797.7	29,751,150	35,696,160.5
6,840 +		97,812 +		35,701,380 +	

It is clear that the ranges regarding the number of animals and nutrients produced is far to wide to provide the public with any kind of meaningful information. It is scientifically inconceivable to claim that whether an operation produces, for example, 38,610 gallons of manure/day versus 52,895.7 gallons of manure/day makes no difference. The proposterousness of the ranges is made even more clear when looking at the difference in manure produced on an annual basis. Without adequate information regarding the number of acres covered by the plan or used for land application of livestock nutrients; livestock nutrients transferred to other persons; and crop yields it is impossible to ascertain whether the waste generated at dairies, AFOs and CAFOs is being contained in a way that protects public health and the environment.

Not only is this information crucial for purposes of the public interest (understanding how these operations affect inherent rights to a clean and healthy environment), but it is also imperative to ascertain whether the agencies are fulfilling their statutory responsibilities to protect public health and the environment from the pollution caused by dairies, AFOs and CAFOs. For example, under the Dairy Nutrient Management Act, Ecology has the authority and obligation to designate a dairy or AFO as a CAFO that discharges to waters of the state (and thus would be required to have a discharge permit) considering the following factors: (a) size of AFO and amount of wastes reaching waters; (b) location of the AFO relative to waters of the

state; and (c) means of conveyance of pollutants to waters of the state. RCW 90.64.020. In essence, this is the same information that is excluded from public review pursuant to WAC 16.06.210(29). Therefore, without that information, the public has no way to ascertain whether Ecology is fulfilling its responsibilities under RCW 90.64.020. That is just one example of many.

Similarly, meaningful information for each of the five categories is critical to citizen participation in enforcement activities regarding how dairies, AFOs, and CAFOs are managed and operated. Without more specific information on the number of animals, quantity of nutrients produced and acreage available for land application, citizens will not be able to understand the amount of waste storage that is available. Nor will they be able to determine the degree to which an expansion of waste loads is feasible, including sufficiency of holding volumes during anticipated precipitation events, without likely environmental contamination or threat to public health. Citizens will also be without information in regards to how the animals are cared for given the size and nature of the operation. Access to all of this information is important for citizens and government regulators alike.

c. The Current Regulation Improperly Expands & Misapplies the Confidential Business Information Exemption

Under the Washington Public Records Act, there is no general exemption for "confidentiality of business information."[64] Rather, the law provides that meaningful

[64] Under the Federal Freedom of Information Act ("FOIA"), there is an exemption (4) that protects "trade secrets and commercial or financial information obtained from a person and privileged or confidential." 5 U.S.C. § 552(b)(4). The Public Records Act does not contain a similar exemption and thus case law interpreting FOIA Exemption 4 should not be relied upon by DOA in determining what constitutes confidential business information under WAC 16.06.210(29). *Francis v. Wash. Dep't of Corrections*, 178 Wash.App. 42, 313 P.3d 457, 465 (2013) ("Washington courts do not consider FOIA cases in interpreting PRA provisions that do not correspond to analogous FOIA provisions.").

information from dairies, AFOs and CAFOs must be disclosed to the public "while ensuring confidentiality of business information . . . ." RCW 42.56.610. The DOA has not defined what constitutes "confidential business information," another fatal flaw in the existing WAC 16.06.210(29).

None of the five categories of information subject to the exemption in WAC 16.06.210(29) constitute confidential business information. Rather, the five categories contain factual data that is critical for the public to access in order to protect the public interest. Even if a dairy, AFO or CAFO operator claimed this information to be confidential, that claim would be outweighed by the public's overwhelming interest in obtaining meaningful information for each of the five categories. As an example, "RCW 43.21A.160 explicitly grants Ecology the ability to determine what constitutes confidential business information." *Comm'y Ass'n for Restoration of the Environment v. Ecology*, 149 Wash.App. 830, 852, 205 P.3d 950 (2009). However, Ecology's authority to grant an exemption for confidential business information is expressly limited. Ecology does not have the authority to identify public information as confidential business information if doing so would be "detrimental to the public interest." RCW 43.21A.160. Any similar attempt by DOA to withhold meaningful information from the public would be constrained by the obligation to release information that is in the public interest.

In addition, any claim of confidentiality would be vitiated by the fact that several dairies, AFOs and CAFOs already share information contained in the five categories to other sources. For example, in regards to chicken CAFOs, egg farmers already regularly share their population information to industry organizations (e.g. the United Egg Producers), trade magazines (e.g. WattAgNet.net and Egg Industry magazine), government agencies like EPA/FDA/SEC, and the news media itself like when a producer wants to build a new farm in a community. The public's need to accurately assess the risk of beef and dairy consumption far outweighs any interests that producers may have in concealing production capacity to protect their commercial interests. In fact, the actual number of animals on farms is far from secret. For instance,

Skyridge Farms in Sunnyside, Washington openly admits to having 3,200 cattle,[65] and shares with competitors production methods that contribute to their success. [66] Given that stocking density is just one of many factors that influences a farm's efficiency, the revelation of granular density data cannot and will not undermine a farm's competitive position. Concealing this information, however, will cripple the ability of consumers and advocates to fully appreciate the risk of beef and dairy consumption in Washington state.

Obtaining this critical information about CAFOs in a patchwork and incomplete manner contravenes the Public Records Act's mandate of free access to information that affects the public interest. Moreover, the likelihood of commercial harm to be non-existent because price competition is not affected by knowledge of competitors' production capacities. Therefore, Commenters urge DOA to revise WAC 16.06.210(29) to make it clear that there are no circumstances under which the factual information contained within the five specific categories would constitute confidential business information.

II. CONCLUSION

The Commenters appreciate the opportunity to comment on the proposed rulemaking regarding the DOA's Numeric Range Rule. We strongly urge DOA to revise WAC 16-16-210(29) to bring it into compliance with the law and so that it provides meaningful information to the public regarding agricultural sources of pollution that affect public health, the environment,

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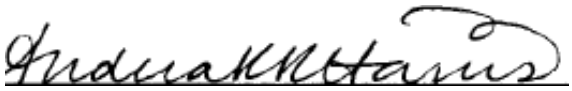
[65] *See, i.e.*, Skyridge Farms, in Sunnyside, WA, discloses that they have 3,200 cows at http://sites.usdairy.com/Sustainability/Awards/Pages/Skyridge-Farm.aspx; notably, according to the U.S. Dairy Sustainability Awards report available for download at the same site, Dan DeGroot, owner of Skyridge Farms, actively shares with others the methods he uses to improve his production.

[66] *See* Jim Barmore, Improving the Bottom-Line – More Cows or Less Crowding, *available at gpsdairy.com/Resources_Docs/ImprovingTheBottom-line.pdf. ("Over-stocking and cow comfort, in many ways, should be considered one of the same." at 3; "Determining an optimum stocking density is really about evaluating the entire housing and milking center as a system." at 9.)*

animal welfare, and that contribute to climate change. Please do not hesitate to contact the undersigned if you have any questions about these comments, or if you would like to discuss any of the issues raised herein further. The undersigned also would like to be added to the list of individuals who receive regularupdates regarding this proposed rulemaking.

Sincerely,



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On Behalf Of: Puget Soundkeeper Alliance, Center for Environmental Law & Policy, Community Association for the Restoration of the Environment, Orca Conservancy, Concerned Citizens of the Yakama Reservation, the Friends of Toppenish Creek; Water & Salmon Committee Sierra Club WA State Chapter, Socially Responsible Agriculture Project

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Via Electronic Mail

June 17, 2014

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Re: Recommendations For Amendment of WAC 16-06-201(29)

Dear Ms. Prest,

In response to your request of April 14, 2014, we are submitting additional recommendations as to how WAC 16-06-210(29) needs to be amended to come into conformance with all applicable law. These recommendations are in addition to, and do not supplant, the comments that we submitted on March 31 and April 14, 2014. We appreciate this additional opportunity to provide recommendations on the Washington Department of Agriculture's ("DOAs") rulemaking to revise the numeric ranges set forth in WAC 16-16-210(29). As we made clear in our initial comments, the Commenters are committed to ensuring that the waters of this state (ground and surface waters) are swimmable, fishable, and sufficiently clean for public consumption. In addition, the Commenters believe that the public has a right to the information they need to protect their interests in a clean and healthy environment for present and future generations and to ensure that government is fulfilling their obligations to protect the public interest, an obligation that has been woefully performed to date.

As we explained in our original comments, the existing ranges provided for in WAC 16.06.210(29) effectively create an exemption for five categories of information, all of which

are crucial pieces of information to ascertain (1) the extent of pollution coming from these operations; (2) whether the government is fulfilling its statutory obligations to protect public health and the environment; and (3) the extent to which these facilities are discharging pollutants into waters of the state without a permit. The five categories of information are (1) Number of animals; (2) volume of livestock nutrients generated; (3) number of acres covered by the plan or used for land application of livestock nutrients; (4) livestock nutrients transferred to other persons; and (5) crop yields. The Legislature has recognized the significance of these categories of information by directing that "meaningful information" about each category be provided to the public. RCW 42.56.610; 90.64.190.

In order to constitute "meaningful information," the Department of Agriculture must release the *actual* number of animals, the *actual* volume of livestock nutrients generated, the *actual* number of acres covered by the plan or used for land application of livestock nutrients, the *actual* amount of livestock nutrients transferred to other person, and the *actual* crop yields. Otherwise, only the CAFOs and the agencies have the information needed to know what is going on and whether these facilities are operating within the bounds of the law. Because CAFOs are point sources under the federal Clean Water Act,[1] all of this information constitutes effluent data[2] that is required to be disclosed under section 308 of the federal Clean Water Act:

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[1] Because all CAFOs are point sources and are discharging to waters of the state either by using lagoons that leak to contain their manure or by over-applying manure to fields, all should be required to be covered under the CAFO General Permit.

[2] "The term 'effluent limitation' means any restriction established by a State or the Administrator on quantities, rates, and concentrations of chemical, physical, biological, and other constituents which are discharged from point sources into navigable waters, the waters of the contiguous zone, or the ocean, including schedules of compliance." 33 U.S.C. § 1362(11); *see also RSR Corp. v. Browner*, 924 F. Supp. 504, 508 (S.D. N.Y. 1996) (citing 40 C.F.R. § 2.302(a)(2)(i)(B)) (noting that EPA Regional Counsel defined "effluent data" as "information necessary to determine the amount, frequency, concentration, temperature, or other characteristics (to the extent related to water quality) of the pollutants which, under an

> Any records, reports, or information obtained under this section (1) shall, in the case of effluent data, be related to any applicable effluent limitations, toxic, pretreatment, or new source performance standards, and (2) *shall be available to the public*, except that upon a showing satisfactory to the Administrator by any person that records, reports, or information, or particular part thereof (*other than effluent data*), to which the Administrator has access under this section, if made public would divulge methods or processes entitled to protection as trade secrets of such person, the Administrator shall consider such record, report, or information, or particular portion thereof confidential in accordance with the purposes of section 1905 of Title 18.

33 U.S.C. § 1318(b) (emphasis added). As you can see from the plain language of the statute, there is no exemption for information claimed to be trade secret or confidential business information. Furthermore, WSDA has presented no justification, nor could it, to show that information contained within the five categories could possibly constitute confidential business information. 40 C.F.R. 2.302(e) (“”[I]nformation which is effluent data . . . is not eligible for confidential treatment” under 40 C.F.R. § 2.208).

As we stated in our original comments, not only is this information crucial for purposes of the public interest (understanding how these operations affect inherent rights to a clean and healthy environment),[3] but it is also imperative to ascertain whether the agencies are fulfilling their statutory responsibilities to protect public health and the environment from the pollution caused by dairies, AFOs and CAFOs. For example, under the Dairy Nutrient Management Act, Ecology has the authority and obligation to designate a dairy or AFO as a CAFO that discharges

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applicable standard or limitation, the source was authorized to discharge (including, to the extent necessary for such purpose, a description of the manner or rate of operation of the source.”).

[3] “The legislature recognizes and declares it to be the policy of this state, that it is a fundamental and inalienable right of the people of the state of Washington to live in a healthful and pleasant environment and to benefit from the proper development and use of its natural resources.” RCW 43.21A.010.

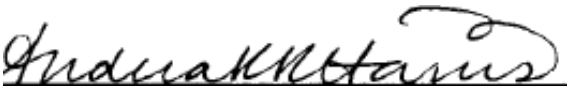
to waters of the state (and thus would be required to have a discharge permit) considering the following factors: (a) size of AFO and amount of wastes reaching waters; (b) location of the AFO relative to waters of the state; and (c) means of conveyance of pollutants to waters of the state. RCW 90.64.020. In essence, this is the same information that is excluded from public review pursuant to WAC 16.06.210(29), but is legally required to be disclosed under the Clean Water Act. Therefore, without that information, the public has no way to ascertain whether Ecology is fulfilling its responsibilities under RCW 90.64.020. *See RSR Corp.*, 925 F. Supp. At 508 (upholding EPA's determination that "the Wallkill Plant's monthly production rate constitute effluent data which is not entitled to confidential treatment . . . .").

Similarly, the actual data for each of the five categories is critical to citizen participation in enforcement activities regarding how dairies, AFOs, and CAFOs are managed and operated. Without the actual and verifiable information on the number of animals, quantity of nutrients produced and acreage available for land application, citizens will not be able to understand whether the facility has waste storage available to contain the vast amount of waste generated. Nor will they be able to determine the degree to which an expansion of waste loads is feasible, including sufficiency of holding volumes during anticipated precipitation events, without likely environmental contamination or threat to public health. Citizens will also be without information in regards to how the animals are cared for given the size and nature of the operation. Access to all of this information is important for citizens and government as is made clear in the federal Clean Water Act.

Again, we appreciate the opportunity to comment on the proposed rulemaking regarding the DOA's Numeric Range Rule. We strongly urge DOA to revise WAC 16-16-210(29) to bring it into compliance with the law and so that it provides meaningful information to the public regarding agricultural sources of pollution that affect public health, the environment, animal welfare, and that contribute to climate change. Claims of trade secret in this context just don't hold water. Please do not hesitate to contact the undersigned if you have any

questions about these comments, or if you would like to discuss any of the issues raised herein further.

Sincerely,



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Emission and Dispersion of Bioaerosols from Dairy Manure Application Sites: Human Health Risk Assessment

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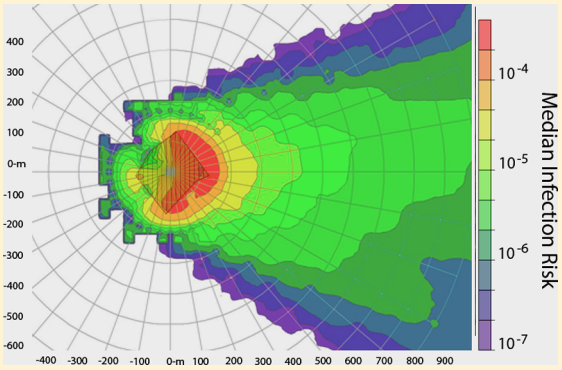
Supporting Information

ABSTRACT: In this study, we report the human health risk of gastrointestinal infection associated with inhalation exposure to airborne zoonotic pathogens emitted following application of dairy cattle manure to land. Inverse dispersion modeling with the USEPA's AERMOD dispersion model was used to determine bioaerosol emission rates based on edge-of-field bioaerosol and source material samples analyzed by real-time quantitative polymerase chain reaction (qPCR). Bioaerosol emissions and transport simulated with AERMOD, previously reported viable manure pathogen contents, relevant exposure pathways, and pathogen-specific dose-response relationships were then used to estimate potential downwind risks with a quantitative microbial risk assessment (QMRA) approach. Median 8-h infection risks decreased exponentially with distance from a median of 1:2700 at edge-of-field to 1:13 000 at 100 m and 1:200 000 at 1000 m; peak risks were considerably greater (1:33, 1:170, and 1:2500, respectively). These results indicate that bioaerosols emitted from manure application sites following manure application may present significant public health risks to downwind receptors. Manure management practices should consider improved controls for bioaerosols in order to reduce the risk of disease transmission.



INTRODUCTION

Application of livestock manure to agricultural land is a conventional practice that provides benefits of waste disposal and crop fertilization. However, concerns of environmental and public health impacts have grown as animal agriculture has become increasingly concentrated.[1,2] The primary health risk associated with manure management is zoonotic pathogens present in land-applied materials.[2,3] While exposure through waterways contaminated by runoff has received considerable attention,[4] the health risk of bioaerosols containing manure pathogens has not been well documented. Nevertheless, epidemiological studies demonstrate excess negative public health outcomes in those residing near large livestock agriculture operations and land application sites.[5,6] Bioaerosols from manure application sites have been cited as an important data gap in understanding their overall risks.[7,8]

The primary human exposure route for bioaerosols containing gastrointestinal pathogens is inhalation followed by deposition in the upper respiratory tract and subsequent swallowing; indirect exposure routes include deposition onto crops or water resources.[7,9] Although several studies have considered emission and transport of bioaerosols from land application of human waste biosolids,[10−13] very few have focused on manure application.[14,15] Manure is typically untreated beyond attenuation in storage lagoons, and is land-applied at high rates compared to human waste biosolids.[2,12] Quantitative microbial risk assessment (QMRA) has been used to synthesize modeling results in order to estimate human health risk; often, data limitations have forced researchers to apply external models and broad assumptions to hypothetical scenarios.[16−18] Of note, manure pathogens persist in soils at high concentrations postapplication[19] and can be resuspended from manure-amended fields.[20] Work to date has focused on emissions during the relatively short application period, and has ignored emissions postapplication that continue over longer time frames.

The objective of this study was to use field-based measurements and QMRA to determine realistic estimates of public health risk downwind of dairy manure application sites (i.e., following application). Edge-of-field bioaerosol samples and applied source materials were analyzed by real-time quantitative polymerase chain reaction (qPCR) in order to determine emission rates of bacteria from land-applied manure by inverse dispersion modeling using the United States Environmental Protection Agency (USEPA) AERMOD (American Meteorological Society/Environmental Protection Agency Regulatory Model) atmospheric dispersion modeling

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system.[21] AERMOD was then used to model emissions and dispersion of bacteria from manure-applied fields. Empirical results formed the basis for QMRA modeling of potential risks to downwind receptors based on previously published distributions of viable manure pathogen contents of cow manure reported by the USEPA.[4] This work provides new data on emission and transport of bioaerosols following manure application, and advances understanding of human health risks associated with manure application practices.

■ MATERIALS AND METHODS

Sample Collection and Analysis. Manure for this study was from a dairy operation in New York State housing approximately 3,200 animals. Cows were fed a mixture of corn grain, silage, and hay, and were bedded on sawdust or sand. Manure was stored for a period of up to six months prior to land application at agronomic rates; manure properties are provided in Supporting Information, SI, Table S1. Field sites included corn (n = 7) and grass (n = 4) crop fields to which manure slurry (mean 5.8% solids) was applied by broadcast application (mobile tanker from a height of 0.5 or 2.7 m) and nearby background fields (n = 6) to which manure was not applied during the previous six months. Samples were collected under a range of conditions (see SI Table S2).

This study aimed to characterize emissions from land-applied manure, rather than from spreading equipment during application as has been considered by other studies.[14,16] Bioaerosol samples were collected at edge-of-field (1.5 m height) for 8 h, commencing immediately after manure application. Sampling equipment included SKC Biosampler liquid impingers (SKC Inc., Eighty Four, PA) for determination of bioaerosol concentration and Andersen-type 6-stage viable impactors (Tisch Environmental, Cleves, OH) for determination of bioaerosol size distribution as previously described.[20] Briefly, phosphate buffered saline was replaced in autoclaved impingers (one per site operated at 12.5 $L \cdot s^{-1}$) every 1 h and composited into a single autoclaved 250 mL bottle; the composite sample was filtered onto a 0.2 μm polycarbonate filter (GE Osmonics, Minnetonka, MN) and stored at −20 °C. Two sets of 6-stage impactor samples (10 min at 28.3 $L \cdot s^{-1}$) were collected per site onto tryptic soy agar and incubated at 25 °C for 5 days. Meteorological data (1 min average wind speed and direction, temperature, and relative humidity) were logged by a Pegasus EX weather station (Columbia Weather Systems, Hillsboro, OR). Manure samples were collected from at least 5 locations on the field, composited into a sterile 50 mL conical tube, and stored at 4 °C (<9 h) until 250 μL aliquots were distributed into sterile 1.5 mL tubes and frozen at −20 °C.

DNA was extracted from impingement samples using the MoBio Ultraclean DNA Isolation Kit (MoBio Laboratories, Inc., Solana Beach, CA) and from manure samples using the MoBio PowerSoil DNA Isolation Kit as per manufacturer instructions, except that bacterial disruption occurred in an MP Biomedicals FastPrep 24 (Irvine, CA; 30 s at 6 $m \cdot s^{-1}$) and salmon testes DNA (Sigma, St. Louis, MO; 2.5 $ng \cdot reaction^{-1}$) was added as an exogenous extraction and amplification control.[22] DNA was stored at −20 °C.

Real-time qPCR assays used in this study (see SI Table S3) included total bacteria, fecal indicator bacteria (*Escherichia coli* and *Enterococcus* spp.), and bacterial pathogens (*Salmonella* spp., *Campylobacter* spp., and *E. coli* O157:H7). Reactions were performed on a Roche LightCycler 480 (F. Hoffmann-La Roche, Ltd., Basel, Switzerland) as previously described.[20] Sample concentrations were determined based on calibration curves generated for each target using serially diluted control DNA. Pathogen gene concentrations ($copies \cdot \mu L^{-1}$) were converted to organism concentrations ($cells \cdot \mu L^{-1}$) based on copies of target genes per cell genome.[23] Refer to SI Table S4 for qPCR calibration curves and limits of detection, and SI Table S5 for details of DNA standards, including gene copies per cell genome. qPCR yields a conservative estimate of viable bacteria as it includes viable, nonviable, and viable-but-not-cultivable (VBNC) cells that remain infectious,[24] and avoids limitations of cultivation including selection bias and underestimation due to sampling stress.[25]

Emission and Transport Modeling. Emission and transport modeling was completed using the USEPA's AERMOD dispersion model.[21] Event-based models were developed in AERMOD View (Lakes Environmental, Waterloo, ON) using ArcGIS (Esri, Redlands, CA) basemap images (Esri World Imagery) and shapefiles of the field sites. Metrological data was from on-site stations and pseudomet station generated from MM5 (Fifth-Generation Penn State/NCAR Mesoscale Model) data (Lakes Environmental). Surface parameters (surface roughness, Bowen ratio, and albedo) were estimated from United States Geological Survey (USGS) National Land Cover Data (NLCD) through the USEPA's AERSURFACE tool. Terrain elevations from USGS 15 min DEM files were processed using the USEPA's AERMAP. Bioaerosol size distribution measured in 6-stage impactors and an assumed density of 1.0 $g \cdot cm^{-3}$ were used to calculate particle dry deposition rates; samples were collected during days with no precipitation therefore no wet deposition occurred. The geometric mean of upper and lower cut points for each stage was used to define size bins.

Edge-of-field fecal indicator bacteria measurements (*Enterococcus* spp.) (C_{obs}; $copies \cdot m^{-3}$) were used to model an 8-h average emission rate (Q_{obs}; $copies \cdot m^{-2} \cdot s^{-1}$) based on inverse dispersion modeling with an arbitrary emission rate (Q_{sim}) and the associated concentration (C_{sim}) simulated by the event-specific AERMOD model, such that,[26,27]

$$\frac{C_{obs}}{Q_{obs}} = \frac{C_{sim}}{Q_{sim}} \quad (1)$$

Emission rates of total fecal bacteria ($copies \cdot m^{-2} \cdot s^{-1}$) were then estimated based on the ratio of *Enterococcus* spp. to total bacteria in the manure. These were converted to emission factors ($copies \cdot L^{-1}$ manure) based on daily application rates; the fraction of bacteria in source material emitted to the air (aerosolization efficiency) was determined based on measured bacterial abundance in the source material. Ambient breathing height (1.5 m) concentrations based on inverse-modeled emission rates were projected in AERMOD at 100 m increments from edge-of-field to 1000 m, at 20 m intervals along each distance. The highest (i.e., directly downwind) modeled bacterial concentrations from each event at 100 m increments were log-transformed and fit to a gamma distribution for each distance.[28] It is important to note that a delivery of total fecal bacteria was calculated based on the ratio of total fecal bacteria concentration in the manure source material to modeled total fecal bacteria concentration in the air as a function of distance from the edge of the manure-amended field. For risk assessment, delivery of viable pathogens in bioaerosols with distance was estimated based on viable pathogen distributions in manure as reported by the USEPA

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Table 1. Input Parameters for Quantitative Microbial Risk Assessment Models

variable	species	model	parameters	values	reference
bacterial air concentration (C_{air}, copies·m^{-3})	total bacteria	gamma	shape	variable[a]	this study
			scale		
bacterial manure concentration (C_{man}, copies·wet g^{-1})	total bacteria	normal	mean	1.11×10^{10}	this study
			SD	5.67×10^{9}	
pathogen manure prevalence (I, %)	*Salmonella* spp.	log-uniform	min.	5	4
			max.	8	
	Campylobacter spp.	log-uniform	min.	5	
			max.	38	
	E. coli O157:H7	log-uniform	min.	9.7	
			max.	28	
pathogen manure abundance (P_{man}, cells·wet g^{-1})	*Salmonella* spp.	log-uniform	log min.	2.6	4
			log max.	4.6	
	Campylobacter spp.	log-uniform	log min.	1.8	
			log max.	4.5	
	E. coli O157:H7	log-normal	log mean	3.08	
			log SD	1.49	
human infectious potential (p, %)	*Salmonella* spp.	point estimate	point estimate	50	4,31
	Campylobacter spp.	point estimate	point estimate	83	
	E. coli O157:H7	point estimate	point estimate	83	
breathing rate (br, m^3·hr^{-1})		log-normal	log mean	0.579	32
			log SD	0.225	
exposure time (t, hr)		point estimate	point estimate	8	
aerosol ingestion rate (ag, %)		uniform	minimum	10	9,11
			maximum	50	
probability of infection (P_I)	Nontyphoid *Salmonella*	β-Poisson	α	0.3126	33
			β	2884	
	Campylobacter jejuni	β-Poisson	α	0.145	34
			β	7.59	
	E. coli O157:H7	β-Poisson	α	0.248	35
			β	48.80	

[a]See SI Table S9 for model distributions at various distances.

in previous QMRA assessments for water quality.[4] Emission of *Enterococcus* spp. and transport of total fecal bacteria were assumed representative of the three bacterial pathogens of this study (scaled by relative manure concentrations), as reported in risk assessment below. The fit of distributions to measured or modeled data were assessed by Shapiro–Wilk and Kolmogorov–Smirnov tests for normal and gamma distributions, respectively. Uncertainty in parameter estimates was included by generating normal or χ^2 observations about estimated values during each Monte Carlo iteration, and the sensitivity of risk estimates to input factors was examined using Spearman's rank order correlation. Statistical analyses were performed using R 3.1.1.[29]

Risk Assessment. Risk assessment was based on USEPA guidelines for microbial risk assessment,[30] a USEPA QMRA of freshwater impacted by agricultural fecal contamination,[4] and published risk assessments of bioaerosols from human waste biosolids application.[10−12,16−18] Relevant parameters in the QMRA model (Table 1) were varied over probability distributions generated during this study, or reported in scientific literature, through Monte Carlo simulation (10 000 iterations) in Vose ModelRisk (Gent, Belgium). Median and peak (95th percentile) risks were considered to represent most likely and potential high-risk scenarios.

Reference pathogens for this study (Table 1) were chosen that are common in livestock manure, persistent in the environment, cause a significant fraction of gastrointestinal illness, and for which dose–response relationships are available in scientific literature.[4,31] Potential downwind exposure was determined based on modeled bioaerosols and typical pathogen contents of cattle manure reported by the USEPA.[4] Data sets forming the basis for the USEPA values included review papers of livestock hosts for pathogens, studies reporting pathogen prevalence and abundance in manures, and studies reporting the occurrence of pathogens as related to geographic area and farm practices.[4] For these bacterial species, the probability of gastrointestinal infection (P_I) given exposure to an effective internal dose (d; cells) is described by a β-Poisson distribution with location and shape parameters α and β derived from the observed dose–response of the particular pathogen:[33]

$$P_I(d) = 1 - \left(1 + \frac{d}{\beta}\right)^{-\alpha} \tag{2}$$

Effective internal dose was determined by coupling airborne pathogen exposure concentration (ec; cells·m^{-3}) with an adult breathing rate distribution (br; m^3·hr^{-1}) and exposure time (t; hr):

$$d = (\mathrm{ec} \times \mathrm{br} \times t) \times \mathrm{ag} \tag{3}$$

where (ag, %) is the fraction of inhaled bacteria that enter the gastrointestinal system; a uniform distribution over the reported range of 10–50% was assumed.[9,11] The exposure concentration of each species was estimated by weighting modeled airborne bacterial concentration (C_{air}, copies·m^{-3}) at each distance by bacterial manure concentration (C_{man}, copies·

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wet g^{-1}), abundance of the pathogen in manure (P_{man}, cells·wet g^{-1}), prevalence of the pathogen in manure (I, %), and human infectious potential (p, %) or fraction of strains present in cattle manure that are also infectious to humans:[16,31]

$$ec = \left(\frac{C_{air}}{C_{man}}\right) \times P_{man} \times I \times p \quad (4)$$

RESULTS AND DISCUSSION

Field Measurements and Inverse Dispersion Modeling. Eight-hour average airborne concentrations of total bacteria measured by liquid impingement at downwind edge-of-field sample points ranged from 1.07×10^5 to 1.35×10^6 copies·m^{-3} (see SI Table S6); mean background was 6.35×10^4 copies·m^{-3} (SD $\pm$ 3.30×10^4 copies·m^{-3}). *Enterococcus* spp. ranged from 1.79×10^2 copies·m^{-3} to 1.54×10^4 copies·m^{-3}; all background samples were below detection limits. This demonstrates that total bacteria concentrations at the downwind edge-of-field may be 0.2−1.3 log greater than background (mean 0.8 log difference) and contain fecal bacteria aerosolized from the field. Bacteria were primarily associated with coarse particles (>2.1 μm; mean 76%; SD $\pm$ 13%; see SI Table S7); size fractioning of cultivable bacteria (i.e., percent of total) was assumed to be representative of total bacteria and fecal pathogens as well. Since the exposure pathway considered here requires initial deposition in the upper respiratory tract, coarse particles are of particular concern.[9]

Mean bacterial abundance in source material was 2.14×10^{11} copies·dry g^{-1} (SD $\pm$ 1.29×10^{11} copies·dry g^{-1}). Mean *Enterococcus* spp. and *E. coli* in the manure were 7.77×10^8 copies·dry g^{-1} (SD $\pm$ 6.76×10^8 copies·dry g^{-1}) and 3.17×10^6 copies·dry g^{-1} (SD $\pm$ 3.41×10^6 copies·dry g^{-1}), respectively. *Campylobacter* spp. was detected in all manure samples (mean 2.25×10^6 copies·dry g^{-1}; SD $\pm$ 3.11×10^6 copies·dry g^{-1}) near the upper range of reported abundances;[4] note that measured concentrations are presented on a dry gram basis and that multiple gene copies are present per cell (equivalent mean concentration 2.53×10^4 cells·wet g^{-1}), and that qPCR yields higher concentrations than cultivation-based measures due to inclusion of nonviable and VBNC cells.[24] *Salmonella* spp. and *E. coli* O157:H7 markers other than *flic*H7 were below limits of detection, within typical abundance ranges.[4] Although *flic*H7 was detectable in all but one sample (mean 2.57×10^5 copies·dry g^{-1}; SD $\pm$ 2.59×10^5 copies·dry g^{-1}), serotypes other than *E. coli* O157 may also express the H7 antigen.[36] *E. coli* and fecal pathogens were not tested in bioaerosol samples due to low manure concentrations; their detection was not anticipated given observed bacterial aerosolization efficiencies and analytical limits of detection. Complete qPCR results are available in SI Table S6.

Since particulate emission rates from ground-level area sources cannot be measured directly, bioaerosols were measured downwind and emissions estimated using inverse dispersion modeling.[26,27] An AERMOD-based model of each site was used to determine the linear relationship between emission rate and concentration at the specific sample point under actual conditions during the event, and thus of an inverse-modeled emission rate that was not sensitive to selection of sample location.[26] Inverse dispersion modeling has been previously applied for manure land application sites,[37] and AERMOD has been used to inverse-model emissions from livestock operations on a similar length scale.[27] Due to the time-integrated nature of impingement sampling, only an average rate over the sampling period (in this case 8 h) could be determined by this method; it is understood that instantaneous emissions varied with meteorological and surface conditions. However, given that these also predict 8-h average concentrations, their use is appropriate in determining risk over the equivalent averaging period.

Eight-hour average bioaerosol emission rates ranged from 7.2×10^3 copies·m^{-2}·s^{-1} to 3.9×10^5 copies·m^{-2}·s^{-1}, equating to 2.5×10^7 copies·L^{-1} to 1.1×10^9 copies·L^{-1} manure over the 8-h period when normalized to application rates; refer to SI Table S8 for emission rates and emission factors for each event. On the basis of the measured bacterial manure abundance during each event, the fraction of bacteria in applied materials emitted to the air (aerosolization efficiency) during the first 8 h following application was 2.8×10^{-6} to 2.7×10^{-4} (mean 6.8×10^{-5} or 0.0068%). While no known prior studies have determined bacterial aerosolization efficiency from manure-applied lands, the values estimated by this study are comparable to those found by Paez−Rubio et al.[13] for total bacteria during application of Class B biosolids (mean 2.5×10^{-5}). Given that most bacteria remained on the field following application, that they survive for extended periods,[19] and that resuspension increases with drier material,[8] additional emissions after the 8-h study period are likely.

Quantitative Microbial Risk Assessment. Downwind concentrations were projected using the AERMOD dispersion model, which has been validated for USEPA regulatory applications.[38] Although designed for assessment of high concentrations in long-term data sets for regulatory compliance (e.g., criteria pollutants on an annual scale), its accuracy at 8-h intervals has been demonstrated and the model has been previously applied for bioaerosol risk assessment.[14,39] Modeled downwind concentrations of bioaerosols for each sample event are provided in SI Table S8. Bioaerosol concentrations at each distance were fit to a gamma distribution using method of moments; this allowed variation in parameter estimates to be nested in Monte Carlo risk simulation models. Use of the empirical distributions, rather than raw measurements for each event, reduced dependency of risk output on individual samples. The gamma distribution has been shown to be an appropriate description of daily bioaerosol concentrations,[28] and model parameters fit well to modeled data (see SI Table S9). Cumulative risk was not sensitive to variability in fitted gamma parameters (Spearman's $\rho \leq 0.03$).

The QMRA scenario in this study coupled bioaerosol emission and transport with manure pathogen contents presented by the USEPA,[4] allowing consideration of generalized potential risks downwind of dairy manure application sites as well as comparison to other QMRA's built upon similar frameworks.[4,14,16] Risks were determined using dose−response relationships previously reported. These models are based on low-dose extrapolation from animal studies or disease outbreaks, and the end point of analysis is infection rather than illness; not all infected individuals will show adverse symptoms.[33] This study is also limited to three reference pathogens, and does not consider other bacterial species, viruses, or protozoa that could increase infection risks.

Prevalence and abundance of pathogens in cattle manure were represented by a log-uniform distribution, with the exception of log-normal *E. coli* O157:H7 abundance due to strong influence of high-shedding events on the overall average.[4,31,40] Although these distributions were used as the

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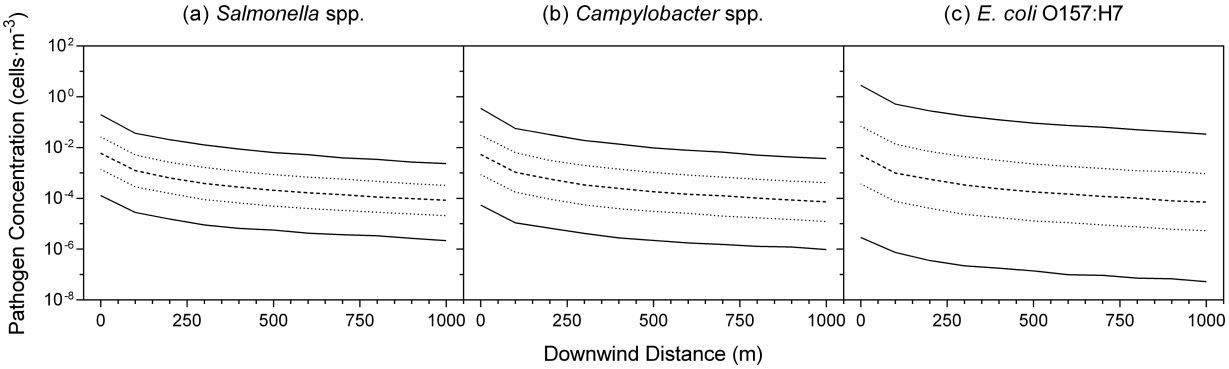

Figure 1. Projected exposure concentrations of (a) *Salmonella* spp., (b) *Campylobacter* spp., and (c) *E. coli* O157:H7 in bioaerosols downwind of manure application sites. Dashed lines indicate median values; dotted lines indicate upper and lower quartiles; solid lines indicate 5th and 95th percentiles.

basis for the USEPA's QMRA of runoff from land application sites,[4] actual pathogen content is influenced by many factors including animal age, diet, bedding, waste handling, herd size, and time of year.[41] Use of probability distributions and Monte Carlo simulation is intended to capture this natural variability. Concentrations are reported for fresh manure rather than slurries, which may be diluted with process wastewaters and/or stored prior to land application, and are combined for both beef and dairy cattle. Following the USEPA's QMRA,[4] abundances are based on reported average (viable) pathogen content rather than that of individual fecal samples in order to represent composite material that is actually land applied.

Figure 1 reports the range of downwind exposure concentrations of pathogens modeled in this study. This study conservatively did not consider decay of pathogens during transport. Although it is known that stress during aerosolization and transport reduces cell viability, quantitative relationships are poorly characterized in the field, and variable with atmospheric conditions, aerosol properties, and bacterial species.[7,14,42] Dungan[14] employed exponential decay rates of 0.002 s^{-1} and 0.07 s^{-1} for low-inactivation and high-inactivation, respectively. Under wind speeds of this study (mean 2.3 $m \cdot s^{-1}$), this equates to low-inactivation survivals of 90% at 100 m and 5% at 1000 m or high-inactivation survival of 40% at 100 m and negligible survival (6×10^{-14}) at 1000 m. Dowd et al.[12] used decay rates of 2.35×10^{-4} s^{-1} and 1.92×10^{-4} s^{-1} for *Salmonella* spp. and *E. coli*, respectively, representing approximately 99% survival at 100 m and 91% survival at 1000 m. However, both sets of decay coefficients were derived from controlled experiments (nebulized pure cultures in a reactor or limited tracer studies with seeded water) that cannot be expected to replicate bioaerosols emitted from manure application sites under variable field conditions.[43] The inconsistency of these assumptions highlights the need for further study of bioaerosol decay in the environment.

On the basis of modeled concentrations shown in Figure 1, median modeled 8-h infection risk decreased exponentially from 3.7×10^{-4} (1:2700) at edge-of-field to 7.6×10^{-5} (1:13 000) at 100 m and 5.1×10^{-6} (1:200 000) at 1,000 m downwind; peak risks were 3.0×10^{-2} (1:33), 5.9×10^{-3} (1:170), and 3.9×10^{-4} (1:2500), respectively (Figure 2). While median exposure concentrations were similar for the three species, potential peak concentrations of *E. coli* O157:H7 were 1-log greater due to the upper tail of its log-normal distribution. As such, while median risk was primarily associated with *Campylobacter* spp. (2.5×10^{-5} at 100 m and 1.7×10^{-6} at

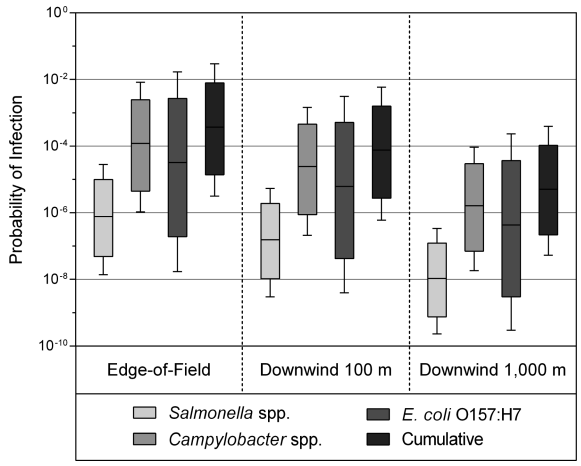

Figure 2. Eight-hour risk of infection from exposure to bioaerosols containing select bacterial pathogens and cumulative (sum) risk at edge-of-field and distances of 100 m and 1000 m downwind. Boxplots indicate upper/lower quartiles and median; whiskers indicate 95th percentiles.

1000 m), peak risk was due to *E. coli* O157:H7 (3.1×10^{-3} and 2.3×10^{-4}, respectively). This indicates that an outbreak of *E. coli* O157:H7 in the herd would create a particularly high-risk scenario worthy of special management considerations, possibly including additional manure treatment and/or restriction of land application. *Salmonella* spp. did not contribute meaningfully to overall risk. Rather, variability was most dependent on *E. coli* O157:H7 and *Campylobacter* spp. manure concentrations ($\rho \geq 0.41$) and modeled bioaerosol concentrations at each distance ($\rho \geq 0.42$), followed by aerosol ingestion rate ($\rho \geq 0.15$). The ingestion rate has not been thoroughly defined and is in this study based on the range assumed in previous work.[9,11] This aspect of the exposure pathway warrants further study. Human infectious potential was treated as a point estimate based on previous studies and did not contribute to variability in risk estimates, although its impact would be anticipated.[4,31]

Two recent QMRAs have considered risks of bioaerosols downwind of manure application, although these studies reported risks associated with bioaerosols generated during application rather than resuspension from manure-amended land following application. Dungan[14] determined risk from exposure during pivot irrigation of dairy wastewater based on pathogen concentrations previously measured using qPCR,[44] a

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range of emission scenarios (wastewater dilutions and aerosolization efficiencies), and dispersion modeled in AERMOD. Low-inactivation infection risks at 1000 m (the closest distance considered) over an 8-h exposure period were substantially greater than those estimated here for the 8-h period following manure application (medium scenario risks of 10^{-3}–10^{-2} for *E. coli* O157:H7 and *Campylobacter jejuni*, 10^{-5}–10^{-4} for *Salmonella* spp.; high scenario risks of 10^{-2}–10^{-1} and 10^{-4}–10^{-3}, respectively). Although overall emission rates (copies·s^{-1}) from the pivot irrigation system and our manure-applied lands were similar, emissions during this study were distributed over large source areas (1.2–10.3 ha) relative to the sprinkler array (0.6 ha), resulting in more disperse downwind concentrations and lower risks (although larger impacted area). This study also included pathogen prevalence and human infectious potential (not all samples were assumed to contain pathogens or strains infectious to humans), which resulted in additional risk reduction. Regardless, both studies predict a rapid decrease in risk with increasing distance from the source.

Brooks et al.[16] included bioaerosols in their comparison of risks during land application of manure versus Class B biosolids. The study used cattle manure pathogen levels reported by Hutchinson et al.[41] (weighted by percent detection), an aerosolization ratio assumed from slinger spreading of dewatered biosolids,[13] and transport models derived for spray tanker application of seeded groundwater.[10] Public infection risks during application (100 m downwind, 1 h duration, 6 days per year) were 1×10^{-8} to 2×10^{-7} for *Salmonella* spp., 5×10^{-7} to 3×10^{-6} for *Campylobacter* spp., and 2×10^{-7} to 8×10^{-6} for *E. coli* O157:H7 (Hutchinson et al.[41] did not report a high peak abundance for *E. coli* O157:H7).

While no guidelines exist for risks associated with bioaerosol exposure, the acceptable risk for fecal contamination of drinking water has been considered 1:10 000;[45] median 8-h risks would fall below this level by 100 m downwind. Conditions associated with high pathogen exposure, including high pathogen manure concentrations and bacterial air concentrations, however, would result in infection risks of 1:2500 at 1000 m downwind. It is important to note that the 8-h infection risks presented in this study are based on exposure to bioaerosols that are resuspended following manure application; additional infection risks will accompany exposure to bioaerosols during application itself, as presented by Dungan[14] and Brooks et al.[16] Further, although an 8-h period was used in this study to coincide with the postapplication monitoring period, exposure will almost certainly continue following the 8-h period of this study, especially for those residing downwind of manure application areas, and increase infection risks. Additionally, instantaneous risks will vary throughout the 8-h period and short-term risks may be greater (although shorter in duration) than the averages presented here. Annual infection risks will exceed the 8-h infection risks and will be an aggregate of the frequency of manure application relative to individual receptors. Moreover, risks to vulnerable subpopulations, including children, the elderly, and immunocompromised individuals, would be greater than those to the general population.[30,33] Epidemiological studies conducted for communities in the vicinity of animal feeding operations and biosolids application sites indicate elevated negative public health outcomes.[5,6] The results of this study and modeling effort demonstrate quantitatively the presence of increased infection risks for those residing near manure application sites. Practices to reduce pathogen contents of manure, such as anaerobic digestion, and restriction of land application activity near residences or public areas would serve to reduce risks of infection and improve public health.

Further research into bacterial viability/survival in bioaerosols, bioaerosol ingestion rates, and dose–response relationships will improve future risk estimates. Other work should address deposition of bioaerosols onto food crops, water bodies, and other relevant surfaces, as well as validate risks in the context of disease incidence, including vulnerable subpopulations such as children. Regardless, this study highlights that bioaerosols emitted from manure application sites present human health risks, and should be considered in a thorough evaluation of livestock manure management practices and their impacts on society and the environment.

■ ASSOCIATED CONTENT

Supporting Information

Additional tables detailing methods and results described in the text. (Table S1) Properties of manures applied in this study and descriptions of the manure storage lagoons; (Table S2) details of field sampling events, including site information, manure application rates, and meteorological data; (Table S3) qPCR assays used by this study and their references; (Table S4) standard curves and performance characteristics for these assays; (Table S5) sources of the DNA standards; (Table S6) qPCR measurements for each event; (Table S7) measured bioaerosol size distributions; (Table S8) results of emission and transport modeling for each event; and (Table S9) downwind bioaerosol concentration distributions modeled across all events. The Supporting Information is available free of charge on the ACS Publications website at DOI: 10.1021/acs.est.5b01981.

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Notes

The authors declare no competing financial interest.

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"It isn't the pollution that's harming the environment, it's the impurities in our air and water that are doing it."

****

Former Vice President Dan Quayle

Bill Dunbar
Policy Advisor
EPA Region 10
dunbar.bill@epa.gov
206/553-1019

Anaerobic digestion, nutrient recovery, biogas…

Improving public health…and bottom lines?

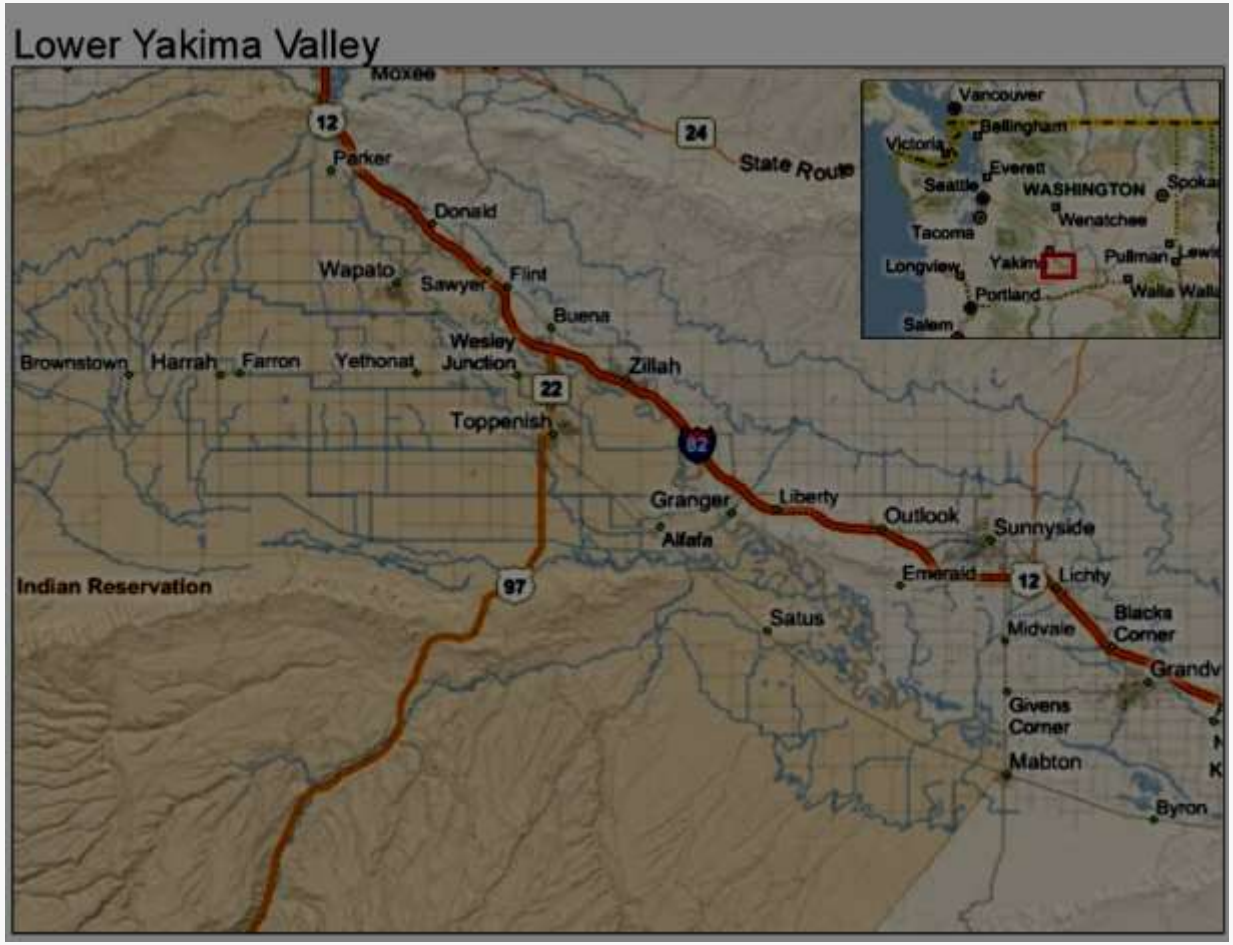
NITRATES…

- **Defined by the SDWA as an "Acute contaminant "**
- Causes methemoglobinemia ("blue baby syndrome") which can be fatal in infants
- **Some studies show cancer association in adults, birth defects**
- Highly mobile in groundwater
- **Maximum Contaminant Level (MCL) for nitrates in drinking water = 10mg/l**
- Documented cases of methemoglobinemia
  - One in Yakima County
  - Five in Benton County

…A problem…

…that's not going away

Lower Yakima Valley



Lower Yakima Valley

- **Yakima County highest childhood poverty level in WA (Census 2009)**
- 75,000 people of diverse backgrounds (40% Hispanic, 6% tribal)
- **Agriculture is primary economic activity**
- ~24,000 people use private drinking water wells
- **More than 2,000 people in study area are using drinking water wells contaminated with nitrate over the MCL**
- 68 dairies in Lower Valley
- **EPA initiated a groundwater nitrates study in response to community concerns about the high nitrate levels in residential drinking water wells and the disproportionate impacts on rural poor populations.**
- Purpose of Study: Identify sources of high nitrate levels in drinking water wells

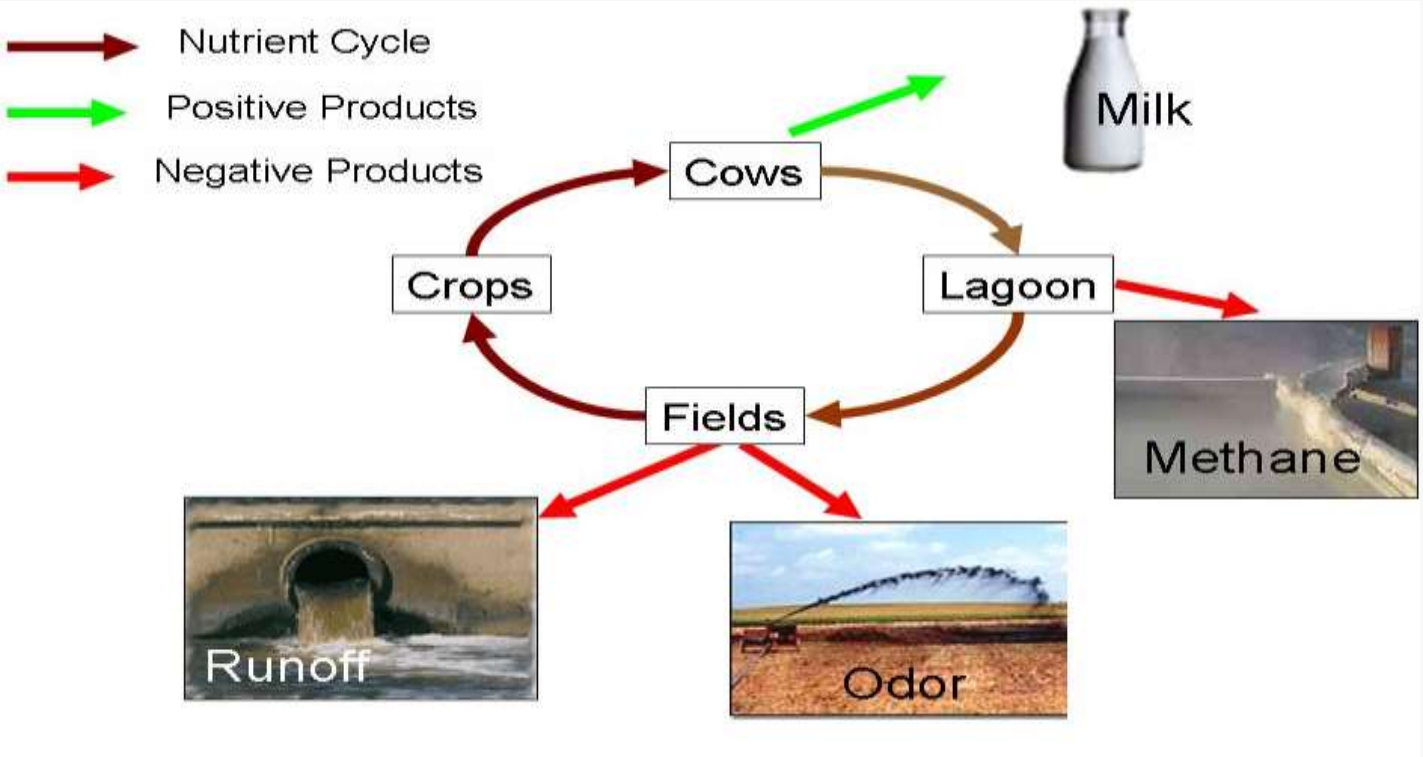
Findings

- » **Dairies** (60% estimated portion of N contribution to groundwater) :
- » Likely source of nitrate in wells down-gradient of them (low levels N up-gradient, high levels down-gradient).

- » **Irrigated cropland** (30%)
- » Likely source based on current and historical use of nitrogen-rich fertilizer – link not as strong as for dairies

- » **Septic systems** (3%):
- » May be a source; study did not confirm

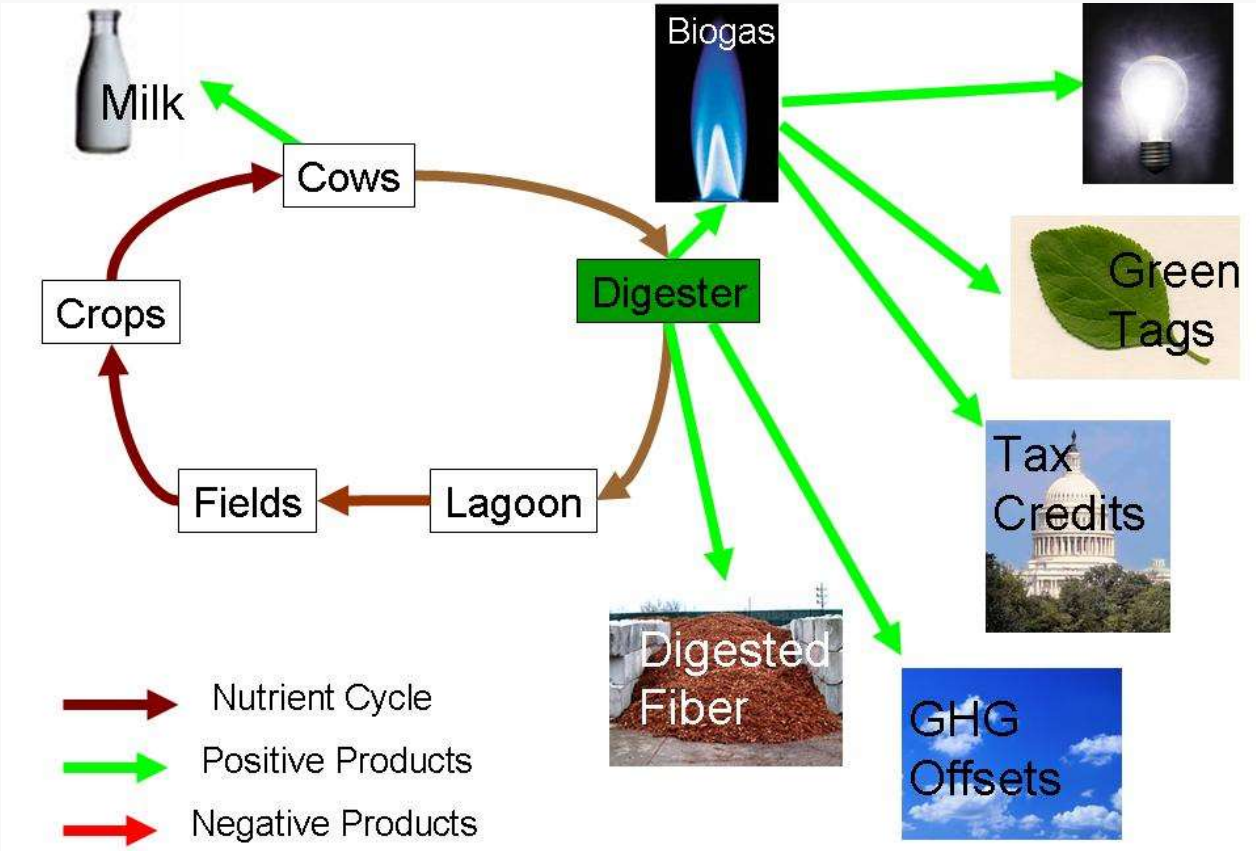
Present Situation



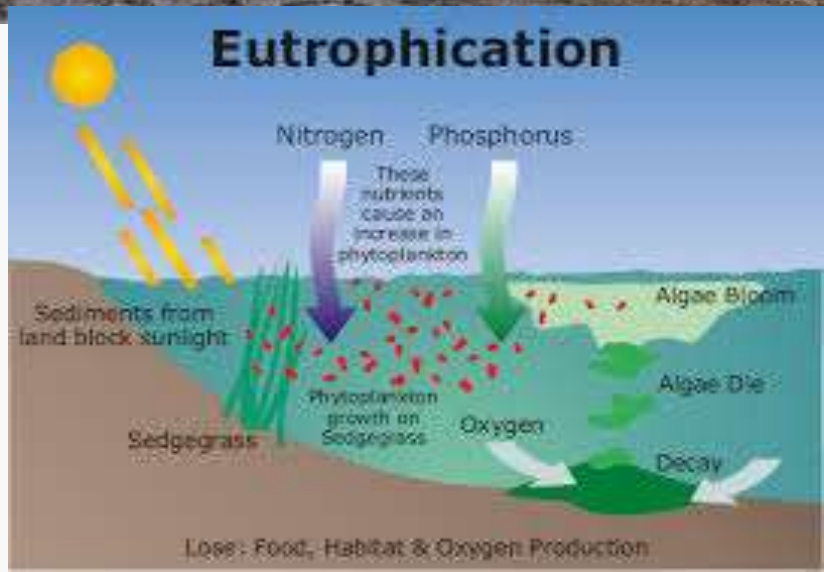
Solids

- Airborne solids from dairies and other livestock feeding operations can cause respiratory problems for downwind neighbors
- On hot, windy days, “dairy dust” (AKA BM-10) can spread pathogens over a wide area.
- Solids from digested manure can replace large amounts of straw and hay dairy farmers purchase as bedding for cows

Anaerobic Digestion



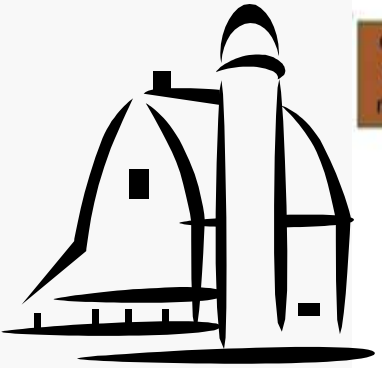
Groundwater contaminated from waste disposal site
Recharge area
Waste disposal site
Water table
Water supply wells
Contaminated groundwater
Environment Canada

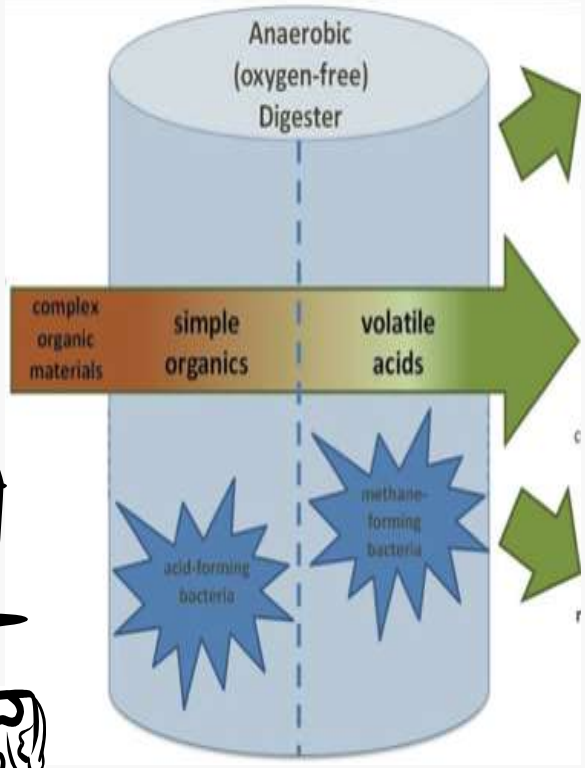

Eutrophication
Nitrogen
Phosphorus
These nutrients cause an increase in phytoplankton
Sediments from land block sunlight
Algae Bloom
Algae Die
Phytoplankton growth on Sedgegrass
Oxygen
Sedgegrass
Decay
Lose: Food, Habitat & Oxygen Production

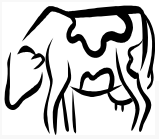
Once again…

Areas at risk of nitrate contamination to shallow ground water

AD mitigates numerous air, water and climate environmental concerns while producing renewable energy **however** little advantage is gained for CAFO or regulatory agencies concerned with nutrient loading to groundwater.







Renewable Energy $
Captured Methane $
Carbon Offsets $

Odor Control
Pathogen Destruction
Solids/Carbon
Destruction

Fibrous Bedding $
Mineralized Ammonia
Mineralized
Phosphorus
NO REDUCTION IN
NUTRIENTS

WASHINGTON STATE UNIVERSITY

Craig Frear, PhD.
Department of Biological Systems Engineering

Commercial Scale NR System -2

- Project site: Vander Haak Dairy, WA
- Capacity: 40,000 gallon/day
- Power Generation: 0.7MW
- Products:
  - ✓ 150 gallon/day $(NH_4)_2SO_4$ solution
  - ✓ 4,000 gallon wet/day P-solid
  - ✓ 36,000 gallon/day low-nutrient effluent

Aeration reactor

Blower

Ammonia Absorption Tower

Ammonium Sulfate Tank

Recovery of Nutrients & Bio-fertilizers

Numerous technologies exist in the municipal sector, some of which are being actively engineered for farm applications, such as that being developed by WSU. Essentially, dilute, soluble forms of nutrients are concentrated and partitioned from the main body of effluent

Dairy Manure AD: Fiber, P-rich fine solids, ammonia sulfate solution

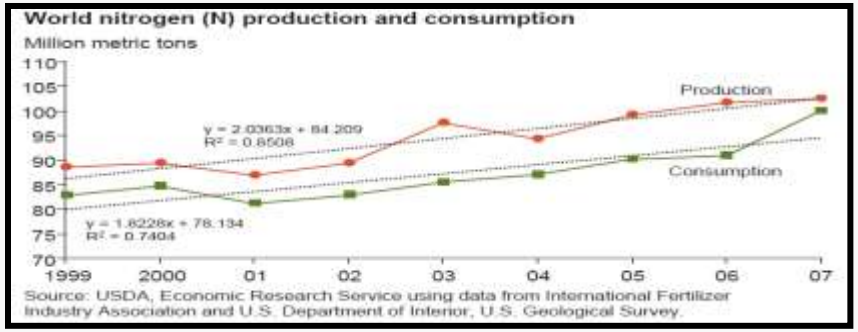
Partitioning and Agronomic Use of Nutrients

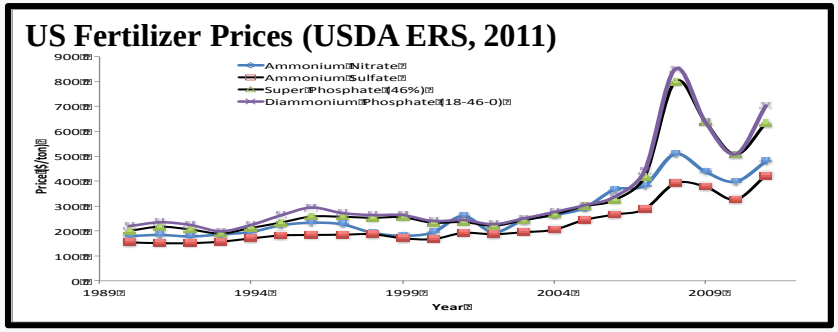
Concentration of nutrients from the effluent through active nutrient recovery systems allows for greater flexibility in producing fertilizer applications suited to particular crops, while also potentially reducing hauling/fueling and ammonia release losses. Most importantly, excess nutrients impacting crops, soil, and waterways/human health can be more effectively exported.

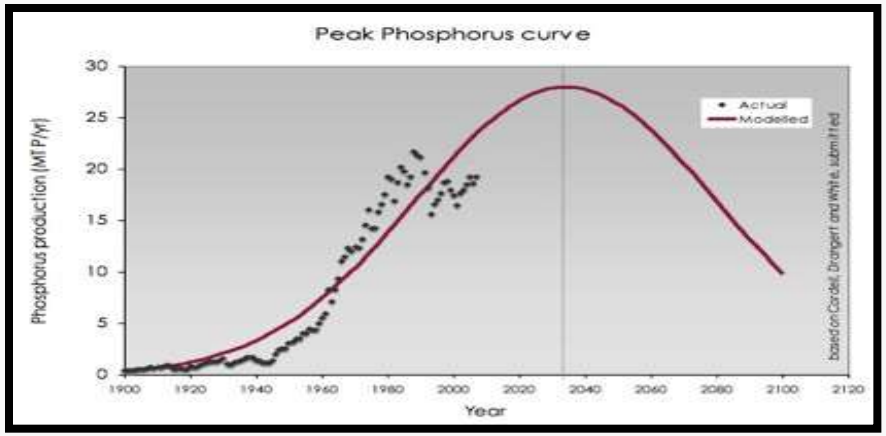
Left: Single pass of manure and ammonium sulfate;
Right: Single pass manure only—same amount of

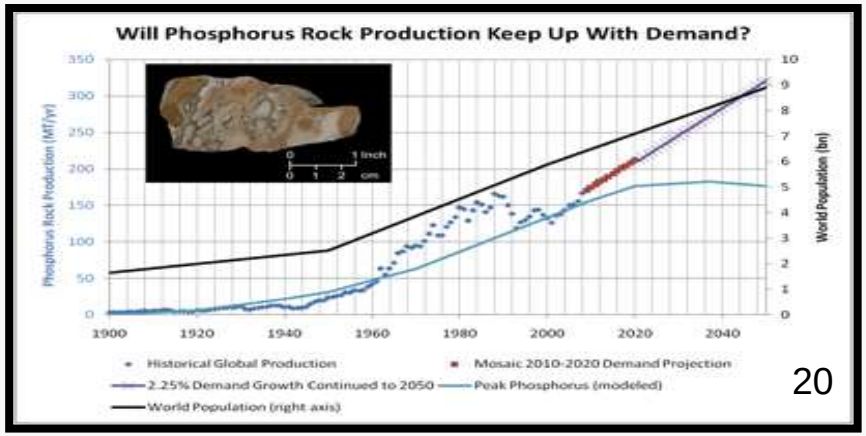
Inorganic Fertilizer Costs

Worldwide consumption of inorganic nitrogen is increasing and nearing present production capacity, while fuel and fertilizer cost parallel each other in elevation. Meanwhile, studies report a future peak phosphorus below world demand.

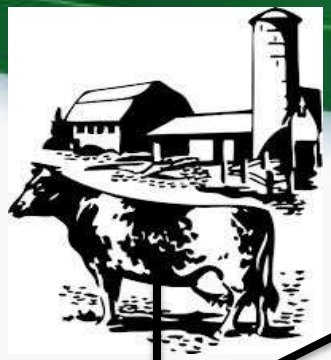








Co-Digestion



239 or 534 KW/year

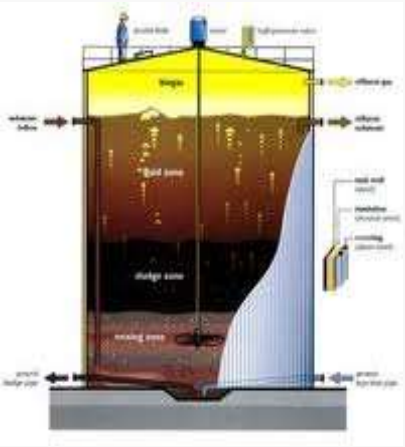
OR CNG

0.6 or 1.3 MGGE/year

Fiber Separation

Ammonia Strip

P-Solids DAF

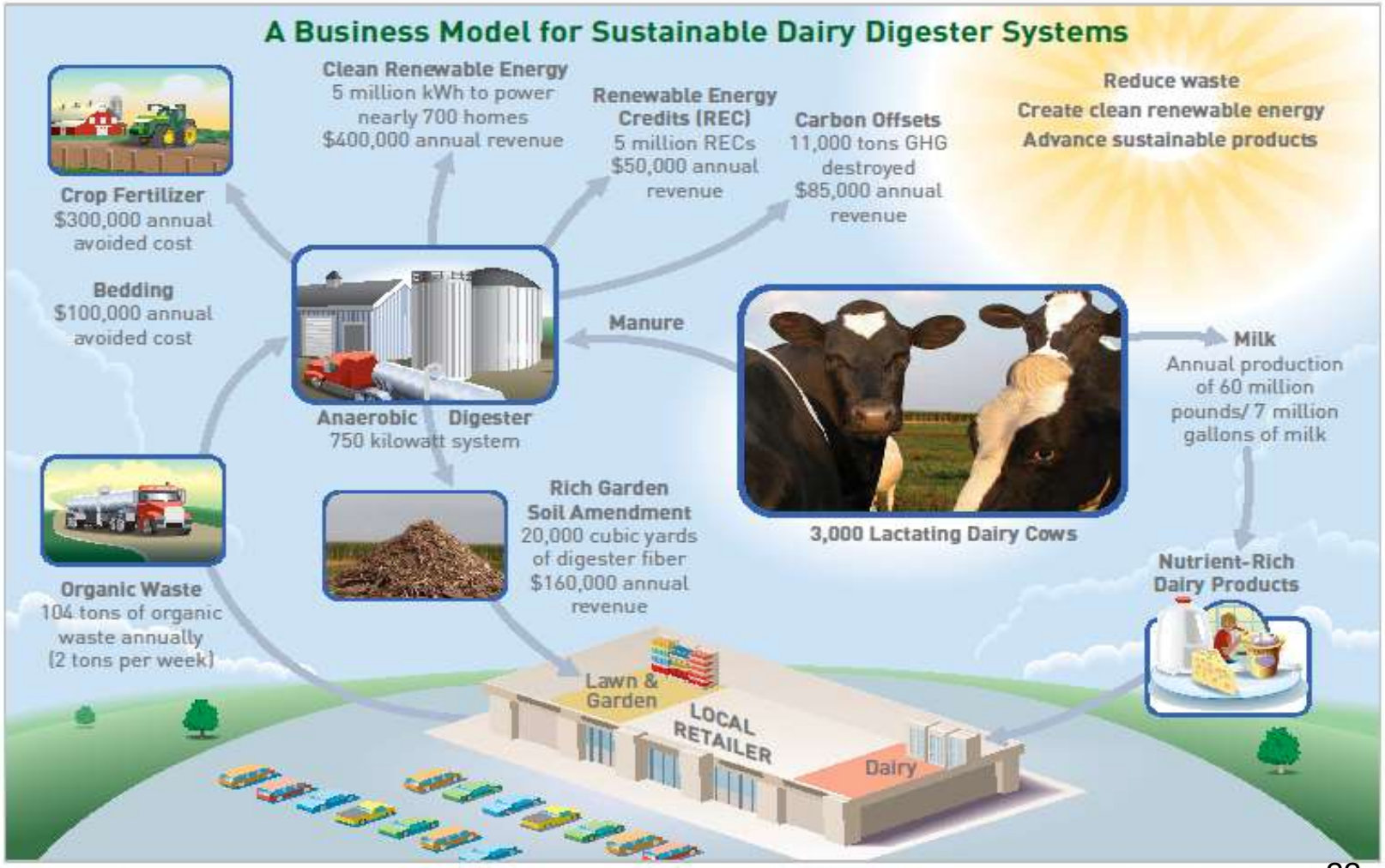


Field Spray

Nutrient Removal CAFO AD/NR System (Blue—1,000 cows/year; Red: 1,000 cows plus 20% v/v co-digestion)

112 tons N 17 tons P	→	100 tons N 13 tons P	→	46 tons N 13 tons P	→	31 tons N 4 tons P
226 tons N 28 tons P	→	202 tons N 24 tons P	→	93 tons N 24 tons P	→	62 tons N 7 tons P

= 72% N Removal & 76% P Removal



For more information go to USDairy.com/Sustainability or contact InnovationCenter@USDairy.com

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U.S. DAIRY SUSTAINABILITY COMMITMENT | INNOVATION CENTER FOR U.S. DAIRY

Back to the Lower Yakima Valley… the Problem Solvers…

- Promus Energy – Dan Evans
- WSU – Craig Frear
- WA Dept. of Commerce – Peter Moulton
- Jim Jensen – WSU Energy
- Mary Beth Lang – WA Dept of Agriculture
- DeRuyter Family
- Dolsen Family
- Bosma Family

"…an interagency methane strategy…"

- **CAFOs**
  - CWA
  - SDWA
  - CAA
  - RNG
  - Valuable N & P, solids
- **WWTPs**
  - CWA
  - CAA
  - RNG
  - Diesel emission reduction
  - Valuable N & P, solids
  - If maximized, significant waste savings
- **Landfills**

“Americans always do the right thing – after they’ve exhausted all other options.”

*****

Winston Churchill

Ecology requires permits, dam safety reviews for high risk dairy lagoons

Dairy lagoons built with uninspected, unpermitted dams can pose a hazard to property and even lives if they fail and cause flooding.

The lagoons are typically used to store dairy waste, also called dairy nutrients, in liquid form over the winter to avoid pollution of ground water or nearby streams, and then the wastewater is dispersed on fields as spray irrigation and as fertilizer during the growing season.

Lagoons are usually constructed with soil embankments (dams) on all four sides. Some are built with common soil embankment dividers producing multi-cell facilities. Portable pumps and temporary pipe may be used for pumping into and out of these lagoons to manually maintain liquid levels.

In just the past year, four unpermitted dairy lagoon dams have been reported to the Department of Ecology by the Natural Resources Conservation Service (NRCS), a branch of the U.S. Department of Agriculture.

MORE INFORMATION

Washington Department of Agriculture Livestock Nutrient Management Program
Phone: 360-902-2894
Web site: http://agr.wa.gov/FoodAnimal/Livestock-Nutrient/Livestocknutrient.htm

CONTACT INFORMATION

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Central Regional Office
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Yakima, WA 98902-3452
(509) 575-2490

Eastern Regional Office
North 4601 Monroe
Spokane, WA 99205-1295
(509) 329-3400

Bellingham Field Office
1440 – 10th Street, Suite 102
Bellingham, WA 98225

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All four are in Whatcom County. Ecology deemed one of them -- found half constructed without permit approval -- a "significant hazard" dam because its failure could potentially flood as many as three residences.

Other safety hazards discovered by Ecology include:

- A 20-years old lagoon found with the spillway filled in with soil to increase its capacity.
- An impoundment built without approved permits that failed once during construction and was rebuilt, again without submittal of a construction permit application, or approval of Ecology's Dam Safety Office.
- Lagoons that had never been inspected and were found in various stages of disrepair including one that had been overfilled and was brimming at the top.

Ecology is seeking voluntary compliance

With 490 dairies identified statewide by the Livestock Nutrient Management (LNM) Program of the Washington Department of Agriculture, other unpermitted lagoons may be improperly designed and constructed.

Ecology's Dam Safety Office has the authority under RCW 90.03.350 and 43.21A.064 to inspect and require permits for lagoons built with more than 10 acre-feet of storage capacity above ground. A lagoon holding 10 acre-feet of dairy waste would be equivalent to a football field, 8 feet deep.

Working in cooperation with Agriculture's LNM Program, Ecology is conducting a statewide inventory of unpermitted dairy lagoons that are large enough to fall under Ecology jurisdiction. Unpermitted jurisdictional lagoons are also being identified through the use of aerial photographs now available for all areas of the state.

Ecology is asking dairy owners to voluntarily bring existing lagoons into compliance with dam safety regulations.

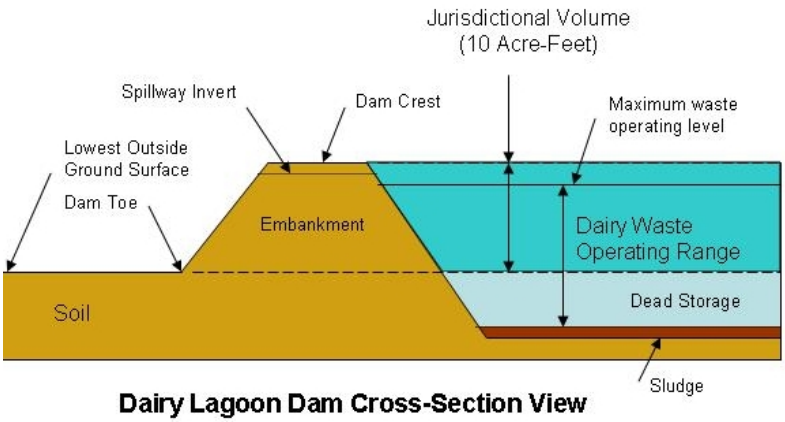
Once a dairy lagoon owner completes any required modifications, and all other needed state permits are obtained, the dam will be in compliance. Owners failing to correct deficiencies could face fines up to $5,000 per day. Ecology can order dangerous lagoons to be drained and removed.

Q: How do I know if my lagoon dam falls under Ecology's authority?

A: If the lagoon impounded behind your dam can store 10 acre-feet (3.26 million gallons) or more above ground level, then it falls under Ecology's authority. This is typically needed on a dairy farm with 200 or more milking cows. This volume is measured from the toe to the crest of the impounding barrier. There is no minimum height for a dam to be under Ecology's authority. A lagoon holding 10 acre-feet of water would be equivalent to a football field, 8 feet deep.

Q: What should I do if I've built or own an unpermitted lagoon that falls under Ecology's authority?

A: Report your dam to Ecology to have it inspected by Ecology engineers who will advise you on how to bring the dam in compliance with state standards.



Dairy Lagoon Dam Cross-Section View

See Dam Safety Office contact information listed at the end of this publication.

Once the lagoon dams have passed an Ecology inspection, all deficiencies have been corrected and you have obtained all other needed permits, your lagoon will be accepted into the Dam Safety inventory and considered a legal facility.

Q: What can I do with my lagoon to avoid it falling under Ecology's authority?

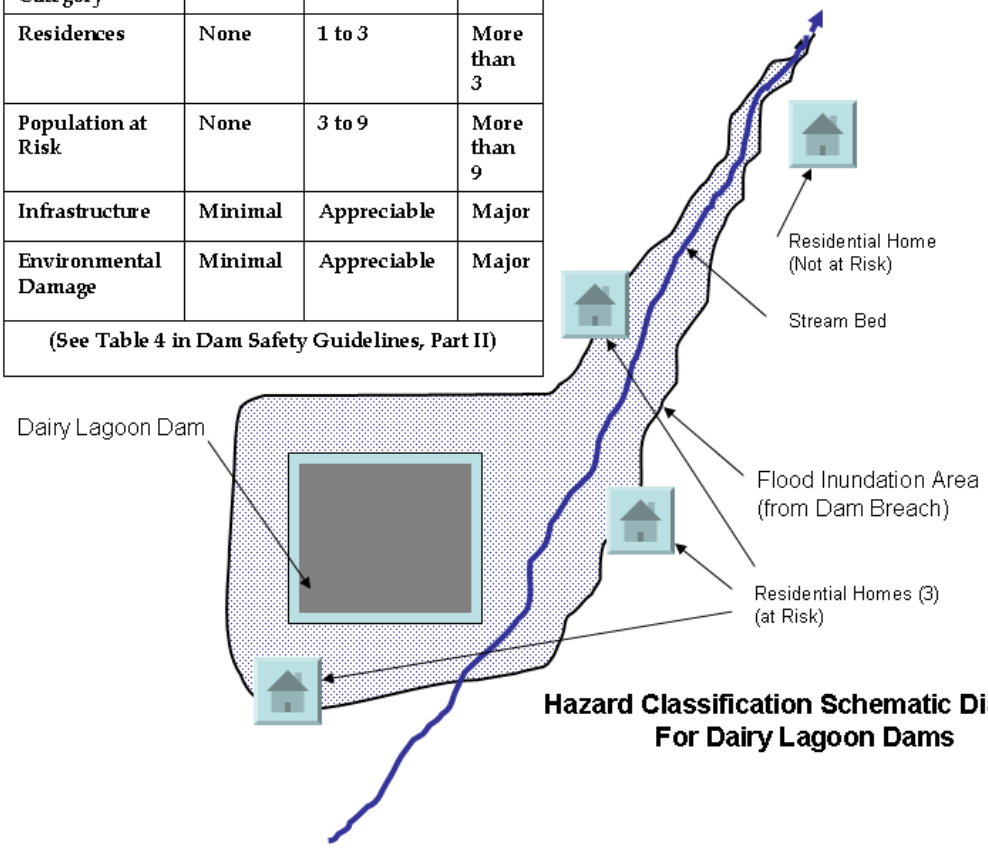
A: One option is to construct or reduce the height of the lagoon dams so they impound less than 10 acre-feet (AF) at dam crest level. Another option would be to excavate the lagoon all or partially below ground level. Only the volume contained between the higher portion of the lagoon floor or the lowest exterior grade at the outer dam toe **and** the dam crest counts towards the 10 AF volume threshold. A third option would be to construct a multi-cell lagoon, with each cell less than 10 acre-feet.

However, even if your impoundment is too small to fall under Ecology's authority, you may be liable for property damage if your lagoon fails. Therefore it is always prudent to have a licensed engineer design any impoundment structure, oversee its construction and periodically inspect it. For existing "low" hazard lagoons (no lives at risk), Agriculture's LNM Program will continue to be the lead inspector for structural adequacy.

Q: How can I determine Ecology's hazard category for my dairy lagoon dam?

A: Ecology uses a hazard category to screen dams for priority for requiring upgrades and enforcement. (See table).

Downstream Hazard Classification			
Hazard Category	Low	Significant	High
Residences	None	1 to 3	More than 3
Population at Risk	None	3 to 9	More than 9
Infrastructure	Minimal	Appreciable	Major
Environmental Damage	Minimal	Appreciable	Major
(See Table 4 in Dam Safety Guidelines, Part II)			



Hazard Classification Schematic Diagram For Dairy Lagoon Dams

Q: Are there any fees associated with owning a lagoon?

A: If your lagoon has a low downstream hazard rating (no lives at risk), then there are no annual fees. However, if your dam has a high (3 or more homes at risk) or significant (1 or 2 homes at risk) downstream hazard rating, then a prorated annual inspection fee is charged, which is adjusted annually for inflation. In 2007, the significant hazard fee was $280, and the high hazard fee was $771. These fees pay for a high hazard dam inspection every 5 years, and a significant hazard dam inspection every 10 years.

Q: What if I want to build a new lagoon that stores 10 or more acre-feet?

A: You will need to follow the standard dam safety permitting process. This would entail hiring an engineer to design the lagoon and prepare plans and specifications. A dam construction permit must be submitted to Ecology along with the plans and a $1400 application fee. Ecology issues a Construction Permit when the construction plans and specifications, and Construction Inspection Plan have been found to be acceptable and upon payment of applicable

plan review fees. Engineering services for the design of dairy waste lagoons are often provided to dairy owners by the U.S. Department of Agriculture's Natural Resources Conservation Service through contracts at the local conservation district.

Q: How long will it take to have my project reviewed?

A: Construction plans and specifications can typically be reviewed and a response letter written to the project engineer within a month after submittal. If it appears the response will take longer than 60 days, the owner or project engineer will be notified in writing and advised of the expected response time.

Q: What is the benefit of bringing my dairy lagoon dam into compliance with Ecology rules?

A: There are several benefits:

1) Avoidance of violations and fines,

2) Assistance from the Dam Safety Office so design and operational standards can be met,

3) For Significant or High hazard facilities, Ecology staff will perform facility inspections to help assure compliance, and

4) Increased protection from legal liability and lawsuits because facility is in compliance with state and industry standards.

Q: How do I get more information on lagoon inspections, dam permits and dam construction?

A: You may contact the Dam Safety Office at:

Department of Ecology
Dam Safety Office
P.O. Box 47600
Olympia, WA 98504-7600

Phone: (360) 407-6883
E-mail: jobl461@ecy.wa.gov

You can visit the Dam Safety Web site at: http://www.ecy.wa.gov/programs/wr/dams/dss.html

DEPARTMENT OF
ECOLOGY
State of Washington

Spreadsheet Models for Determining the Influence of Land Applications of Fertilizer on Underlying Groundwater Nitrate Concentrations

July 2014

Publication No. 14-03-018

Publication and contact information

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Spreadsheet Models for Determining the Influence of Land Applications of Fertilizer on Underlying Groundwater Nitrate Concentrations

by

Charles F. Pitz, Licensed Hydrogeologist

Environmental Assessment Program
Washington State Department of Ecology
Olympia, Washington 98504-7710

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Abstract

Groundwater quality characterization studies have identified significant regional-scale problems with nitrate contamination across Washington State. This contamination is often found in close association with nonpoint applications of nitrogen-bearing fertilizers or animal manure to agricultural lands. Due to the risk that nitrate poses to state drinking water supplies, determining the proper balance between nutrient application rates, crop uptake, and nitrate loss to groundwater is a growing priority in Washington.

This report presents a set of three spreadsheet computer models that can be used to quantitatively predict the impact of residual or excess farm-field soil nitrate on the concentration of nitrate in underlying shallow groundwater.

- Model 1 (NO3-LEACHATE) is a mixing model used to predict the dissolved nitrate concentration that results when a known volume of recharge infiltrates and fully mixes with a residual mass of nitrate present in vadose-zone soils.

- Model 2 (GWNO3-FORECAST) is a mass balance mixing model used to predict the groundwater nitrate concentration that will result when the nitrate-bearing leachate predicted by Model 1 enters an aquifer and mixes with ambient groundwater flowing beneath the site of application.

- Model 3 (GWNO3-BACKCAST) uses modified versions of the Model 1 and 2 equations to back-calculate the nitrate concentration in vadose-zone leachate, the nitrate mass load, and the average shallow soil nitrate concentration required to maintain a given concentration of nitrate in underlying groundwater.

All three models allow the user flexibility in defining site-specific model variables, including the subsurface hydrologic properties of the site, estimates of soil nitrate concentration and bulk density, and nitrate attenuation processes active in both the vadose and saturated zones. Model assumptions and limitations are discussed.

The purpose of this publication is to provide guidance to users of these models.

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Introduction

Contamination of groundwater with dissolved nitrate (NO_3^-; a highly soluble anion containing nitrogen) is a widespread problem in Washington State. Groundwater quality characterization studies have identified significant regional-scale problems with nitrate, including aquifer systems in the central Columbia Basin, the Yakima Basin, and Whatcom County (Ryker and Frans, 2000; Sell and Knutson, 2002; USEPA, 2013; Carey and Cummings, 2012). Wells sampled during these studies have shown dissolved nitrate concentrations frequently above (failing) the Washington State groundwater quality standard of 10 milligrams per liter (mg/L) nitrate as nitrogen (nitrate-N or NO_3-N)(Chapter 173-200 WAC).

Although nitrate can be derived from a variety of natural and anthropogenic sources, nonpoint applications or releases of nitrogen-bearing fertilizers and animal manure to agricultural lands represent the largest sources of nitrate mass released to the environment (Puckett, 1994; Viers et al., 2012). In several areas of concern in Washington, nitrogen fertilizer[1] applications to agricultural lands have been identified as the primary known or likely source of nitrate contamination in groundwater (Frans, 2000; Carey and Cummings, 2012; USEPA, 2013; Carey and Harrison, 2014). As a result, the proper management and control of nitrogen related to agricultural activities is becoming a subject of increasing focus for the Washington State Department of Ecology and for the agricultural industry at large.

Due to its high solubility, nitrate present in a soil column in amounts that exceed crop demand can be rapidly mobilized into solution by downward infiltrating recharge derived from precipitation, irrigation, or both. When the resulting nitrate-enriched leachate moves below the base of the crop root zone, the nutrients are no longer available for plant uptake. If this leachate continues to move downward through the vadose zone and reaches the water table, it can pose a significant risk to the quality of the underlying groundwater. Due to this risk, determining the proper balance between nutrient application rates, crop uptake, and nitrate loss to groundwater is a growing priority in Washington.

This report documents a set of three mathematical models that can be used to quantitatively predict the influence of residual or excess farm-field soil nitrate on the concentration of nitrate in underlying groundwater. The models are based on adaptations of standard mixing and mass-balance principles that have been used to examine contaminant loading to groundwater for many decades (e.g., Summers et al., 1980; Wehrmann, 1984; Bauman and Shafer, 1985; USEPA, 1996; Taylor, 2003; Viers et al., 2012). The models have been incorporated into an interactive Microsoft Excel 2007® spreadsheet file issued with this report.

The tools described in this report are intended to be used to:

- Assist decision-making about nitrogen fertilizer application rates at the field scale.
- Improve the user's understanding of the potential consequences to underlying groundwater quality when land-applying nitrogen-enriched fertilizer in excess of crop demand.

[1] The term *fertilizer* includes both inorganic and organic (manure-based) forms.

- Identify the factors that play the most significant role in nitrate loading impacts to groundwater systems.
- Compare the relative potential for nitrate groundwater contamination between sites.

The models presented in this report allow the user to specify the values for many of the equation terms; this flexibility makes it possible to generate predictions using site-specific conditions. Model assumptions and limitations are presented for each model to guide the user in deciding appropriate use and in assigning appropriate uncertainty to the predictions. Model users are expected to have a working understanding of the sciences of hydrogeology and nitrogen cycling.

Model 1 - NO3-LEACHATE

NO3-LEACHATE Model Description

The NO3-LEACHATE model assumes that all recharge[2] reaching the water table during a model scenario period of interest dissolves (fully mixes with) all residual nitrate mass present in the vadose zone[3] beneath the modeled field. The model calculates the dissolved nitrate-N concentration of the resulting leachate arriving at the water table. The model can accommodate ten independent model scenarios.

Estimates of the residual nitrate-N mass available to leach can be derived from either: (A) near-surface soil nitrate-N sampling results, or (B) a farm-field nitrogen mass balance analysis. Any dissolved nitrate-N present in the recharge prior to its infiltration into the subsurface is accounted for in the final leachate concentration prediction. The model can also account for any nitrate concentration decrease that occurs in the leachate due to attenuation processes active during mixing and transport within the vadose zone.

A schematic of the conceptual model for Model 1 is shown in Figure 1. Variable symbols shown on the figure are explained in detail below. The predicted Model 1 leachate concentration value for each model scenario is automatically carried forward for use as an input variable in the corresponding Model 2 scenario (GWNO3-FORECAST; described later in this report).

[2] For this report, recharge is defined as the water volume that infiltrates through the vadose zone, reaches the water table, and adds to groundwater storage by the end of the model scenario period of interest. Recharge can be derived from the infiltration of precipitation, or from irrigation, or from a combination of the two.

[3] For this report, the *vadose zone* is defined as the zone between the land surface and the regional water table, including the soil column.

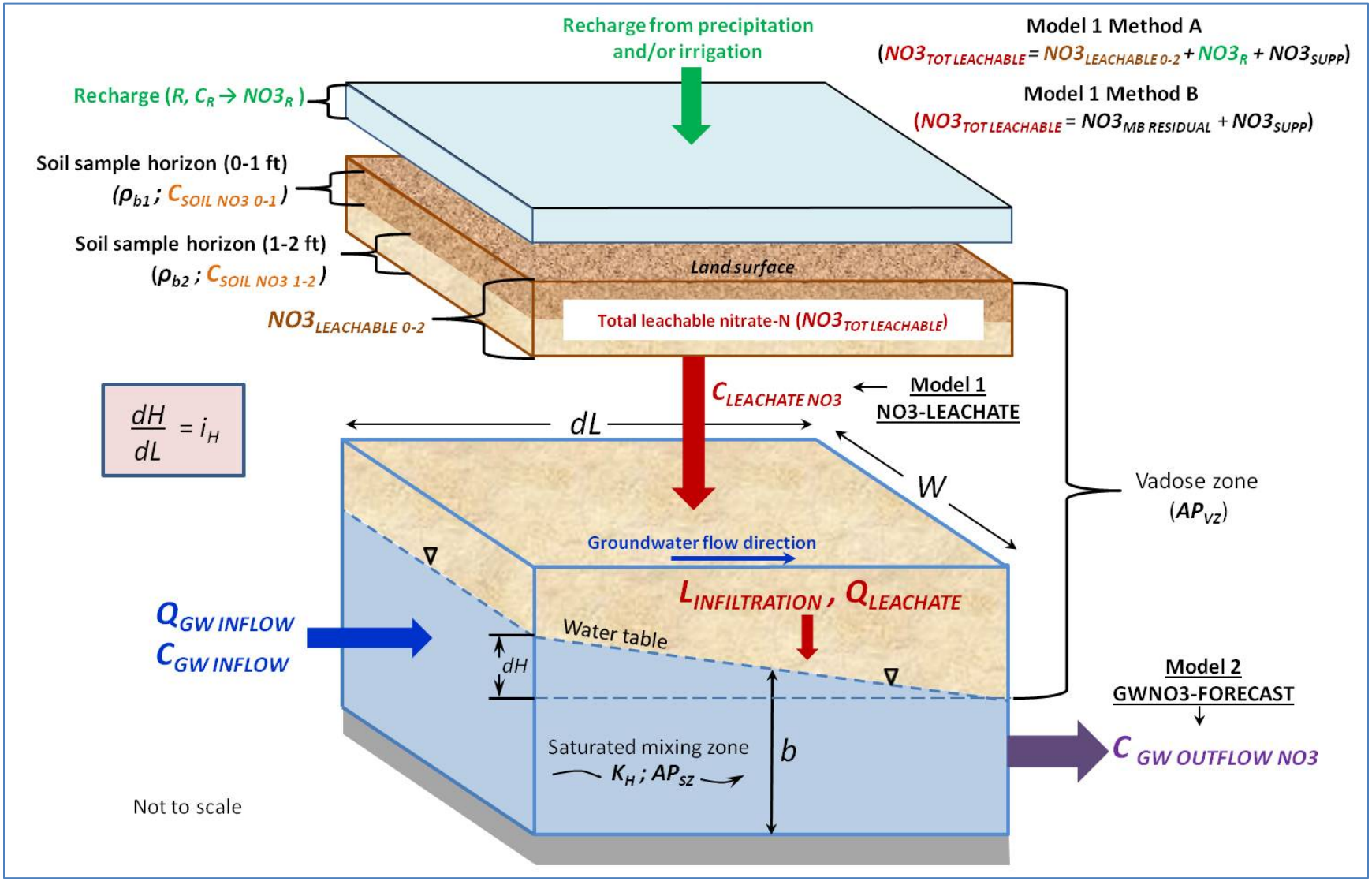

Figure 1. Conceptual Model Schematic – Models 1 (NO3-LEACHATE) and 2 (GWNO3-FORECAST).
Refer to text for further explanation of figure symbols.

In the NO3-LEACHATE model, two different approaches can be used to estimate the amount of residual nitrate-N available for dissolution:

NO3-LEACHATE, Method A

In Method A, the mass of residual nitrate-N available to leach to groundwater from the upper soil column is calculated using soil nitrate-N sample concentration data (or estimates) from the 0-1 and 1-2 foot soil horizons ($C_{SOIL\ NO3\ 0\text{-}1}$ and $C_{SOIL\ NO3\ 1\text{-}2}$). It is a common agronomic practice on fertilized fields to collect soil nitrate-N samples from the uppermost portion of the soil column at the end of the growing season (see for example Sullivan and Cogger, 2003; Ehrhardt and Bundy, 1995; Camberato et al., 2013; Dinkins and Jones, 2007; Carey, 2002; Carey and Harrison, 2014). This type of field data therefore provides a readily available basis for estimating leachable soil nitrate mass.

The total leachable soil nitrate-N mass estimate for the top two feet of the soil column ($NO3_{LEACHABLE\ 0\text{-}2}$) is determined by first converting the average measured (or assumed) soil nitrate *concentration* value to an equivalent nitrate-N unit area *mass* for each horizon, and then summing these two mass values. Soil bulk density values assigned to each horizon by the user ($\rho_{b\ 0\text{-}1}$ and $\rho_{b\ 1\text{-}2}$) are factored into the concentration-to-mass conversion.

In addition to the $NO3_{LEACHABLE\ 0\text{-}2}$ mass, the Method A model can, if appropriate, also account for two additional sources of nitrate-N mass not accounted for by the $NO3_{LEACHABLE\ 0\text{-}2}$ term:

- The nitrate-N mass contributed by the recharge itself ($NO3_R$). To determine the $NO3_R$ value, the user provides (1) the estimated nitrate-N concentration of the recharge *prior to its entry* into the vadose zone ($C_{R\ NO3}$) and (2) the total volume of recharge (R) estimated to reach the water table by the end of the scenario period of interest. For each scenario, the model then combines these values to calculate the mass of nitrate-N supplied by infiltrating precipitation and/or irrigation water.
- A mass of vadose-zone nitrate-N that mixes with the infiltrating recharge that is *supplemental to* the $NO3_{LEACHABLE\ 0\text{-}2}$ value ($NO3_{SUPP}$). This variable allows the user to account for soluble nitrate-N that is (or becomes) available to leach during each model scenario period of interest that is *not otherwise* accounted for by the $NO3_{LEACHABLE\ 0\text{-}2}$ value. Example nitrate sources the $NO3_{SUPP}$ variable could represent include:
    - Leachable nitrate-N mass present in the vadose zone below 2 feet.
    - Nitrate-N mass leached *prior* to soil nitrate sampling (but still within the scenario period of interest).
    - Nitrate-N mass produced by continued microbial conversion of organic nitrogen to nitrate (by mineralization/nitrification), occurring after soil nitrate sample collection, but still within the scenario period of interest. (Norton, 2008; UCD, 2009).

The three distinct sources of soluble nitrate-N mass ($NO3_{LEACHABLE\ 0\text{-}2}$, $NO3_R$, and $NO3_{SUPP}$) are summed to determine the total nitrate-N mass available to leach to groundwater per acre ($NO3_{TOT\ LEACHABLE}$).

NO3-LEACHATE, Method B

In place of soil sampling results, this method uses the residual from a mass balance evaluation of nitrogen inputs and outputs to a farm field ($N_{MB\ RESIDUAL}$) to estimate the mass of soluble nitrate-N available for leaching per acre. It is assumed that the *nitrogen* mass residual estimated by the mass balance is fully converted to *nitrate* and is therefore equivalent to the mass of soluble nitrate-N available for leaching within the scenario period of interest (i.e., $N_{MB\ RESIDUAL} = NO3_{MB\ RESIDUAL}$).

Similar to Method A, Method B also allows the user to assign an additional mass contribution from supplemental sources of nitrate-N ($NO3_{SUPP}$) for any reason not otherwise accounted for by the mass balance analysis, if appropriate. The nitrate-N contribution from recharge is assumed to already be accounted for in the mass balance input-output evaluation and is therefore not included as a separate, explicit variable in the Method B equations.

The two distinct sources of soluble nitrate-N mass estimated for Method B ($NO3_{MB\ Residual}$ and $NO3_{SUPP}$) are summed to determine the total nitrate-N mass available to leach to groundwater per acre ($NO3_{TOT\ LEACHABLE}$).

NO3-LEACHATE, Methods A and B

Once a $NO3_{TOT\ LEACHABLE}$ mass value has been determined for a model scenario (using either Method A or B above), the value is combined with the scenario period-of-interest recharge estimate (R) to calculate the nitrate concentration of the leachate reaching the water table. Before predicting the final leachate concentration, the NO3-LEACHATE model allows the user to also account for any vadose-zone processes (such as denitrification) that act to decrease the nitrate concentration of the leachate during mixing and transport. This is accomplished by applying a term, *vadose-zone attenuation percentage* (AP_{VZ}), in the final model calculation.

Method A and/or B model predictions of the nitrate-N concentration of the leachate are automatically graphed for each model scenario by the accompanying Excel 2007® worksheet (Figure 2).

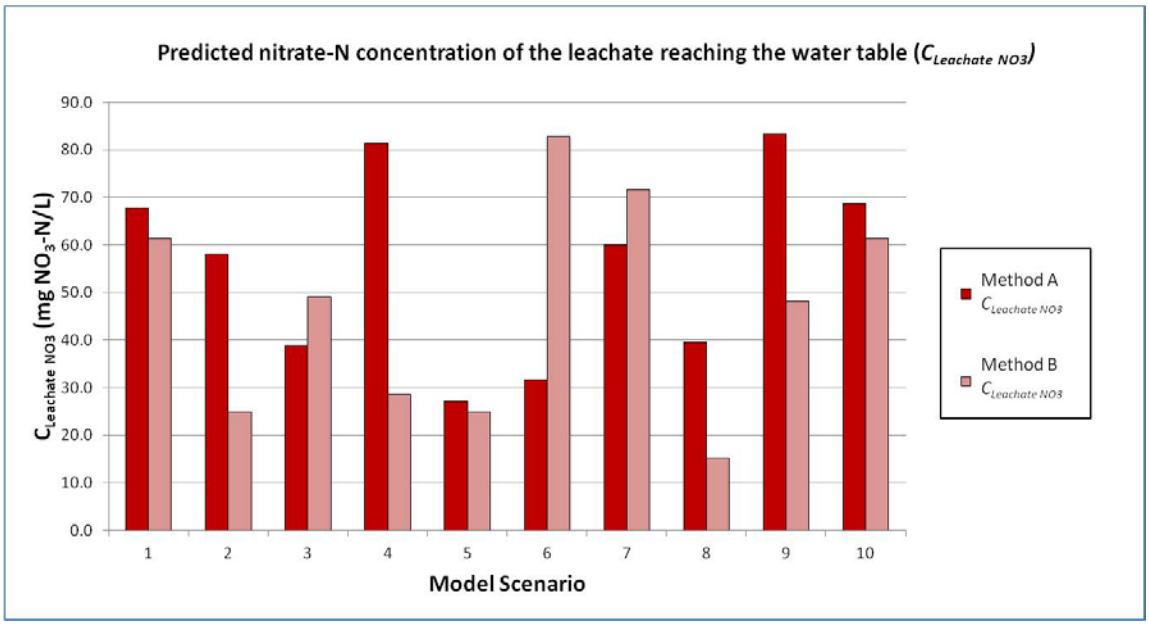

Figure 2. Example output graph from the NO3-LEACHATE model spreadsheet.

NO3-LEACHATE Model Equations

If using Method A

Near-surface soil nitrate-N sample concentration data can be used to estimate the mass of leachable nitrate-N in the top 2 feet of the soil column for each model scenario using Equation 1:

$$NO3_{LEACHABLE\ 0-2} = NO3_{LEACHABLE\ 0-1} + NO3_{LEACHABLE\ 1-2} \qquad \text{Eq. 1}$$

where:

$$NO3_{LEACHABLE\ 0-1} = 2.719(\rho_{b\ 0-1} * C_{SOIL\ NO3\ 0-1}) \qquad \text{Eq. 2}$$

and:

$$NO3_{LEACHABLE\ 1-2} = 2.719(\rho_{b\ 1-2} * C_{SOIL\ NO3\ 1-2}) \qquad \text{Eq. 3}$$

where:

$NO3_{LEACHABLE\ 0\text{-}2}$ = leachable nitrate-N mass present in the 0-2 foot soil horizon (lbs NO_3-N/acre).

$NO3_{LEACHABLE\ 0\text{-}1}$ = leachable nitrate-N mass present in the 0-1 foot soil horizon (lbs NO_3-N/acre).

$NO3_{LEACHABLE\ 1\text{-}2}$ = leachable nitrate-N mass present in the 1-2 foot soil horizon (lbs NO_3-N/acre).

$\rho_{b\ 0\text{-}1}$ = average soil bulk density, 0-1 ft soil horizon (g/cm^3).

$C_{SOIL\ NO3\ 0\text{-}1}$ = average dry-weight soil nitrate-N concentration in the 0-1 foot soil horizon (mg NO_3-N/Kg DW).

$\rho_{b\ 1\text{-}2}$ = average soil bulk density, 1-2 ft soil horizon (g/cm^3).

$C_{SOIL\ NO3\ 1\text{-}2}$ = average dry-weight soil nitrate-N concentration in the 1-2 foot soil horizon (mg NO_3-N/Kg DW).

The total mass of nitrate-N that is available to leach to groundwater is calculated as follows:

$$NO3_{TOT\ LEACHABLE} = NO3_{LEACHABLE\ 0-2} + NO3_R + NO3_{SUPP} \qquad \text{Eq. 4}$$

where:

$$NO3_R = 2.719(R * C_{R\ NO3}) \qquad \text{Eq. 5}$$

where:

$NO3_{TOT\ LEACHABLE}$ = total nitrate-N mass available to leach to groundwater (lbs NO_3-N/acre).

$NO3_R$ = mass of soluble nitrate-N contributed to the leachate by recharge (lbs NO_3-N/acre).

$NO3_{SUPP}$ = supplemental nitrate-N mass contributed to the leachate that is *not otherwise* accounted for by the $NO3_{LEACHABLE\ 0\text{-}2}$ or $NO3_R$ values (lbs NO_3-N/acre).

R = amount of recharge reaching the water table during the scenario period of interest (ft).

$C_{R\ NO3}$ = nitrate-N concentration in recharge (precipitation and/or irrigation water) *prior to* infiltration (mg NO_3-N/L).

If using Method B

Users relying on a farm-field mass balance approach can calculate $NO3_{TOT\ LEACHABLE}$ for each model scenario using a mass balance residual value in the following manner:

$$NO3_{TOT\ LEACHABLE} = NO3_{MB\ RESIDUAL} + NO3_{SUPP} \qquad \text{Eq. 6}$$

assuming:

$$NO3_{MB\ RESIDUAL} = N_{MB\ RESIDUAL} = N_{INPUTS} - N_{OUTPUTS} \qquad \text{Eq. 7}$$

where:

$NO3_{MB\ RESIDUAL}$ = the residual nitrate-N mass determined from a mass balance analysis of nitrogen inputs and outputs to the model area (lbs NO_3-N/acre).

$NO3_{SUPP}$ = supplemental nitrate-N mass contributed to the leachate that is *not otherwise* accounted for by the mass balance analysis (lbs NO_3-N/acre).

$N_{MB\ RESIDUAL}$ = farm-field mass balance nitrogen residual (lbs-N/acre).

N_{INPUTS} = sum of total nitrogen inputs to the model area (lbs-N/acre).

$N_{OUTPUTS}$ = sum of total nitrogen outputs from the model area (lbs-N/acre).

Method A and B

Once a $NO3_{TOT\ LEACHABLE}$ value has been determined, the nitrate-N concentration of the leachate reaching the water table ($C_{LEACHATE\ NO3}$ on Figure 1) can be estimated for each model scenario using a modified version of an equation presented by Carey and Harrison (2014):

$$C_{LEACHATE\ NO3} = \frac{NO3_{TOT\ LEACHABLE}}{2.719R} * (1 - \frac{AP_{VZ}}{100}) \qquad \text{Eq. 8}$$

where:

$C_{LEACHATE\ NO3}$ = nitrate-N concentration of the leachate that reaches the water table (mg NO_3-N/L).

AP_{VZ} = vadose-zone attenuation percentage; percent nitrate-N concentration decrease that occurs in the leachate during transport through the vadose zone, due to denitrification or other attenuation processes (0% to 100%).

See Appendix A for the derivation of the conversion factors used in the above equations.

NO3-LEACHATE Model Assumptions

The model calculations assume the following conditions for each scenario:

- The model predictions represent the condition at the end of the scenario period of interest.
- The soil properties $C_{SOIL\ NO3\ 0\text{-}1}$, $C_{SOIL\ NO3\ 1\text{-}2}$, $\rho_{b\ 0\text{-}1}$, and $\rho_{b\ 1\text{-}2}$ are spatially uniform throughout the model area and depth-intervals of interest.
- The vadose zone is isotropic and homogeneous throughout the model area; vadose-zone attenuation effects on leachate concentrations act equally throughout the vadose-zone volume of interest, and downward flow is uniform.
- The full volume of recharge reaches the water table by the end of the scenario period of interest.
- The concentration of nitrate-N in soil determined using standard laboratory analysis techniques (e.g., 2.0N KCL extraction/ cadmium reduction method; Galvak et al., 2005) is assumed to be equivalent to the concentration of nitrate-N available to leach via recharge infiltration (Horneck, 2014).
- The leachable nitrate-N mass estimated for each scenario is fully dissolved in, and fully mixes with, the infiltrating recharge by the end of the scenario period of interest.
- Dissolved nitrate is unaffected by adsorption within the vadose zone (i.e., the sorption distribution coefficient (K_d) for nitrate is assumed to be 0 ml/g).
- For leachate predictions based on farm-field mass balance nitrogen residuals, it is assumed that the residual nitrogen will entirely convert to soluble nitrate-N within the scenario period of interest, and that nitrate-N mass contributions from recharge are already accounted for in the $NO3_{MB\ RESIDUAL}$ value. The model does not account for errors or uncertainty in the nitrogen mass balance analysis itself.
- Model scenarios are treated independently of one another and are not necessarily connected in time. Model scenarios do not inherit the adjacent scenario's input-variable values; instead, initial conditions for each scenario must be set by the user.

NO3-LEACHATE Model User-Defined Variables

The following variables can be defined by the user for each scenario in the NO3-LEACHATE model:

- $\rho_{b\ 0\text{-}1}$ and $\rho_{b\ 1\text{-}2}$ – these values may be derived from site-specific field testing. Alternatively, Appendix B presents soil bulk density ranges, broken down by soil texture class and depth-interval (0-1' and 1-2'), for both eastern and western Washington soils used for cropland. These ranges were derived from analyses of information in the U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS) SSURGO database (Campbell, 2014).

  Spreadsheets of the raw SSURGO data that were compiled for the analysis (including the NRCS soil map unit) are available as supplemental files to this report. Model users who have knowledge of the specific soil survey map unit(s) at their site (see the NRCS Web Soil Survey website at: http://websoilsurvey.nrcs.usda.gov/) may use this information to support site-specific soil bulk density determinations.

- $C_{SOIL\ NO3\ 0\text{-}1}$ and $C_{SOIL\ NO3\ 1\text{-}2}$ – if using Method A, these values are ideally derived from site-specific soil sampling results. The values should represent the average soil nitrate-N concentration for each depth horizon, across the entire area of interest being modeled (see for example standard soil nitrate sample collection procedures described by Sullivan and Cogger, 2003). In some areas, soil nitrate sampling is only conducted for the top one foot of the soil column. Users in this case can enter an assumed concentration value for the $C_{SOIL\ NO3\ 1\text{-}2}$ term. If this term is ignored (i.e., set to a value of 0 mg/Kg), any soil nitrate present in the 1-2 foot horizon will be ignored in the model calculations.

- $NO3_{SUPP}$ – if a site-specific value for this variable is unavailable, a default value of 0 lbs/acre is recommended for the initial model run. In some cases, this value may only be estimated through an inverse calibration process with Model 2 (described below). The resulting $NO3_{SUPP}$ value may generically represent nitrate-N loading from a source or sources that are otherwise difficult to accurately quantify in the field (e.g., post-sampling mineralization/nitrification).

- R – this value should preferably be derived from a site-specific analysis of recharge. The value should represent the total amount of recharge, from all sources (precipitation and/or irrigation) estimated to reach the water table by the end of the model scenario period of interest. Healy (2010) provides in-depth discussion of the methods available to estimate recharge. If an annual rate of recharge is used to determine an R value for a model scenario that represents less than a full year, R should be decreased accordingly (and seasonal variability should be accounted for). Users should also be sure that the chosen value for R is reasonably consistent with the model assumptions regarding complete mixing and dissolution. For example, even if the recharge value set by the user for the scenario is very small, the model will nonetheless assume that that water volume will completely mix and flush the designated leachable nitrate-N mass. Although this is an unlikely occurrence in the field, the model will nonetheless proceed with the calculations, resulting in an unrealistically high $C_{LEACHATE\ NO3}$ prediction.

- $C_{R\ NO3}$ – for recharge derived solely from precipitation, a recommended value for this variable for sites located east of the Cascade Mountains is 0.09 mg NO_3-N/L. West of the Cascades, a value of 0.05 mg NO_3-N/L is suggested (see Appendix C). If recharge is partly or wholly derived from irrigation, the nitrate-N concentration of the irrigation water prior to application should be measured and accounted for in the $C_{R\ NO3}$ value.
- $NO3_{MB\ Residual}$ – if using Method B, this value should be derived from a site-specific mass balance analysis of farm-field nitrogen inputs and outputs. If the model user has evidence to suggest that only a portion of the total nitrogen residual identified by the mass balance analysis converts to soluble nitrate (i.e., $N_{MB\ RESIDUAL} \neq NO3_{MB\ RESIDUAL}$), the user should adjust the model input value for $NO3_{MB\ RESIDUAL}$ accordingly (see UCD, 2009 and Sullivan, 2008 for additional discussion). If the mass balance nitrogen residual is a negative value, a value of zero (0) should be entered into the model for this variable.
- AP_{VZ} – if a site-specific value for this variable is unavailable, a default value of 0% is recommended for the initial model run (i.e., conservative transport). Model users should note that field research has indicated that concentration reductions of dissolved nitrate-N during transport between the base of a root zone and the capillary fringe of the water table can be very limited in many settings. When this does occur, it can be a highly localized phenomenon (Green et al., 2008; Onsoy et al.; 2005; Artiola, 1997). Holden and Fierer (2005) also note the tendency for denitrification rates to drop off rapidly with depth in the vadose zone. Users setting the AP_{VZ} value above ~10% should have compelling field information to justify the higher value.

NO3-LEACHATE Model Limitations

The NO3-LEACHATE model greatly simplifies the highly complex nature of chemical dissolution and transport observed in many vadose-zone settings. For example:

- Time is not explicitly accounted for in the NO3-LEACHATE model equations; the mixing of recharge with the leachable nitrate mass present within the vadose zone, and the transport of the resulting solution to the water table, is simply assumed to be complete by the end of each model scenario. When interpreting the model results, users should keep in mind any special field conditions at their site that may differ significantly from this assumption (e.g., long-term storage of infiltration in the vadose zone, a deep water table, or delays in downward infiltration due to permeability contrasts).
- The model assumption that the designated recharge volume will fully mix with and dissolve <u>all</u> of the nitrate-N mass present in the soil column within the scenario period of interest ignores the potential influence of spatial heterogeneity, anisotropy, and preferential (non-uniform) flow. In the field, however, these factors can cause large variations in the extent, timing, and rate of nitrate dissolution and transport (see, for example, Sebilo et al., 2013; Onsoy et al., 2005).
- The model ignores any processes occurring during the scenario period of interest such as plant uptake of soil nitrate that may reduce the amount of nitrate mass that is available to leach. This suggests that the NO3-LEACHATE model is best used for examining nitrate leaching outside of the active growing season.

- The accuracies of the NO3-LEACHATE model predictions are a function of uncertainties or errors in the leachable nitrate mass estimates derived by soil sampling (Method A) or farm-field mass balance analysis (Methods B). For example, Carey and Harrison (2014) demonstrated that the timing of soil nitrate sample collection can be a critical factor when attempting to evaluate the amount of soil nitrate available for leaching to groundwater. These authors showed that mineralization processes active subsequent to soil sampling may produce leachable soil nitrate mass not accounted for by the sample results. Shallow soil sampling may also fail to measure nitrate mass that has moved deeper into the vadose zone.

  Similarly, nitrogen mass balance analysis methods normally rely on assumptions about complex nitrogen cycle processes, such as mineralization or volatilization rates, that may not accurately reflect the modeled site.

  Failure to account for all components of the nitrogen cycle, or errors in the measurement or estimation of those components, can impact the accuracy of the residual nitrate mass estimates required by the NO3-LEACHATE model (in turn, influencing the accuracy of the $C_{LEACHATE\ NO3}$ predictions).

Assuming the estimate(s) of total leachable nitrate accurately reflects the nitrate mass actually present in the subsurface at the modeled site, the assumption of *complete* dissolution and transport of that mass may tend to overestimate the amount of nitrate mass reaching the water table during a scenario. In that case, the model predictions of the leachate concentration should be considered conservative (upper-bound) estimates. However, if the nitrate mass available to leach within the model domain is underestimated, the leachate concentration predictions may be lower than what actually occurs in the field.

Model 2 - GWNO3-FORECAST

GWNO3-FORECAST Model Description

The GWNO3-FORECAST model estimates the nitrate-N groundwater concentration that will result after a nitrate-enriched leachate enters an aquifer from the overlying vadose zone and mixes with ambient groundwater flowing beneath an application site.

In this mass-balance box model, based on principles described by Summers et al. (1980), leachate arriving at the water table with a nitrate concentration $C_{LEACHATE\ NO3}$ enters a saturated mixing zone of interest with a user-defined average thickness b (see Figure 1). The leachate fully mixes with groundwater that is entering the upgradient boundary of the mixing zone with a nitrate concentration of $C_{GW\ INFLOW\ NO3}$ at a rate of $Q_{GW\ INFLOW}$.

The model predicts the steady-state nitrate-N concentration of the resulting groundwater/leachate mix that exits the downgradient boundary of the model domain ($C_{GW\ OUTFLOW\ NO3}$). If appropriate, the user can account for processes that act to decrease the groundwater nitrate-N concentration during mixing and transport (for example, saturated-zone denitrification). This is accomplished by applying a term called the saturated-zone attenuation percentage (AP_{SZ}) in the final model calculation. The AP_{SZ} variable is intended to represent attenuation processes that are *distinct from* those that occur within the vadose zone.

To calculate the $C_{GW\ OUTFLOW\ NO3}$ value, the GWNO3-FORECAST model uses a $C_{LEACHATE\ NO3}$ concentration value carried forward from the corresponding scenario from the NO3-LEACHATE model. The GWNO3-FORECAST predictions for $C_{GW\ OUTFLOW\ NO3}$ are automatically graphed by the spreadsheet for a maximum of ten independent model scenarios.

In the accompanying spreadsheet file, the user can also enter measured concentrations of nitrate-N in groundwater for each period to determine how well the model predictions align with field-based data (Figure 3). If appropriate, the model can be calibrated to the field data by adjusting the user-designated values; this calibration process *can include modification of the Model 1 inputs*. If the model is calibrated using this inverse approach, input values should only be adjusted within a range deemed suitable for the site conditions, or consistent with results from previous research.

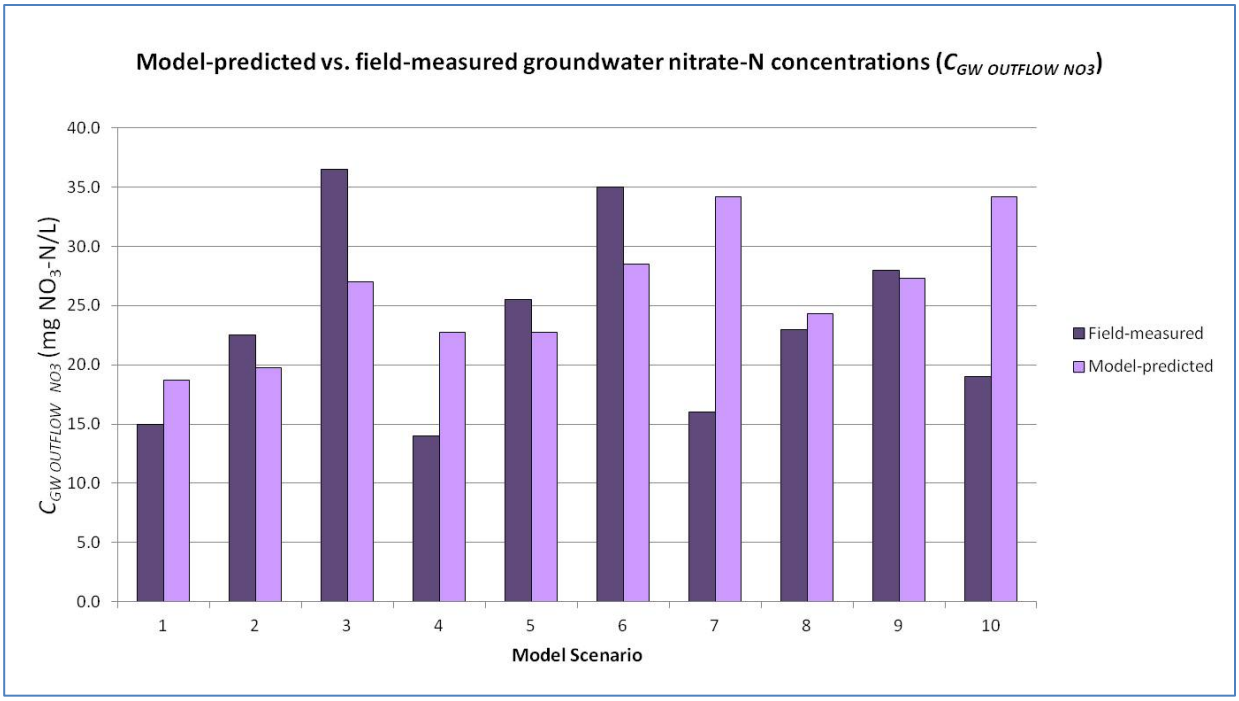

Figure 3. Example output graph from the GWNO3-FORECAST model spreadsheet.

GWNO3-FORECAST Model Equations

Equation 9, modified from Summers et al. (1980) and KGS (1994), is used to estimate the groundwater nitrate-N concentration resulting from the infiltration and mixing of a nitrate-enriched leachate into an underlying saturated zone:

$$C_{GW\ OUTFLOW\ NO3} = \frac{(Q_{LEACHATE} \times C_{LEACHATE\ NO3}) + (Q_{GW\ INFLOW} \times C_{GW\ INFLOW\ NO3})}{Q_{LEACHATE} + Q_{GW\ INFLOW}} * \left(1 - \frac{AP_{SZ}}{100}\right)$$

Eq. 9

where, for each model scenario:

$C_{GW\ OUTFLOW\ NO3}$ = steady-state groundwater nitrate-N concentration exiting the downgradient boundary of the saturated mixing zone at the end of the scenario period of interest (mg NO_3-N/L).

$Q_{LEACHATE}$ = volumetric flow rate of leachate entering the saturated mixing zone from the vadose zone (L/day).

$Q_{GW\ INFLOW}$ = volumetric flow rate of groundwater inflow entering the upgradient boundary of the saturated mixing zone (L/day).

$C_{GW\ INFLOW\ NO3}$ = concentration of nitrate-N in upgradient groundwater inflow *prior to* entering the saturated mixing zone (mg NO_3-N/L).

AP_{SZ} = saturated zone attenuation percentage; percent nitrate-N concentration decrease that occurs in the groundwater/leachate mix during transport through the saturated mixing zone, due to denitrification or other attenuation processes (0% to 100%);

where: $$Q_{LEACHATE} = 28.32(dL * W * L_{INFILTRATION}) \quad \text{Eq. 10}$$

and: $$Q_{GW\ INFLOW} = 28.32(K_H * b * W * i_H) \quad \text{Eq. 11}$$

where: $$i_H = \left(\frac{dH}{dL}\right) \quad \text{Eq. 12}$$

where:

dL = length of the modeled study area, parallel to the groundwater flow direction (ft).

W = horizontal width of the modeled study area, perpendicular to the groundwater flow direction (ft).

$L_{INFILTRATION}$ = infiltration rate of recharge-derived leachate into the saturated mixing zone (ft/day).

K_H = bulk horizontal hydraulic conductivity of the mixing zone aquifer material (ft/day).

b = average thickness of the saturated mixing zone of interest (ft).

i_H = horizontal hydraulic gradient across the modeled study area (ft/ft).

dH = decrease in hydraulic head over the distance dL(ft; entered as a *positive* value).

See Appendix A for the derivation of the conversion factors used in the above equations.

GWNO3-FORECAST Model Assumptions

The GWNO3-FORECAST model calculations assume the following conditions for each model scenario:

- The model predictions for $C_{GW\ OUTFLOW\ NO3}$ represent a steady-state condition at the end of the scenario period of interest.
- The saturated zone of interest is unconfined.
- The saturated zone of interest is isotropic and homogeneous; therefore, mixing and attenuation effects on groundwater concentrations act uniformly throughout the mixing zone. The user-assigned saturated zone attenuation is complete by the end of the scenario period of interest.
- In the accompanying spreadsheet model file, the predicted values for $C_{LEACHATE\ NO3}$ are automatically carried forward from Model 1 (NO3-LEACHATE) to the corresponding Model 2 scenarios. The spreadsheet requires the model user to choose either the Model 1 Method A or Method B $C_{LEACHATE\ NO3}$ values.
- The leachate entering the mixing zone <u>fully</u> mixes with the groundwater estimated to enter the upgradient boundary of the model during that scenario.

- Dissolved nitrate transport through the saturated zone is unaffected by adsorption reactions (i.e., the nitrate sorption distribution coefficient (K_d) is assumed to be 0 ml/g).
- Model scenarios are treated independently of one another and are not necessarily connected in time. Therefore, scenarios do not inherit the adjacent scenario's input-variable values. Initial conditions for each scenario must be set by the user.

GWNO3-FORECAST User-Defined Variables

The following variables can be defined by the user for each scenario in the GWNO3-FORECAST model:

- $C_{LEACHATE\ NO3}$ – In the accompanying spreadsheet model file, the value for this variable is automatically carried forward from the corresponding NO3-LEACHATE model scenario[4]; the user can choose either the Method A or Method B predictions for this purpose.

  If the GWNO3-FORECAST predictions for $C_{GW\ OUTFLOW\ NO3}$ are not in sufficiently close agreement with the concentrations measured in groundwater at the site, the user may consider adjusting some or all of the assigned Model 1 or Model 2 input values to improve the Model 2 calibration. For example, model predictions for $C_{GW\ OUTFLOW\ NO3}$ that are significantly below field measurements of groundwater nitrate concentration suggest that more nitrate mass may be reaching the aquifer than initially predicted. Possible calibration responses to this situation include increasing the $NO3_{SUPP}$ value in Model 1, or decreasing the attenuation percentages used in either Model 1 or Model 2. As mentioned earlier, input values should be modified only within the range deemed suitable for the site conditions, or consistent with results from previous research.
- dL and W – These horizontal dimensions of the modeled area of interest (parallel and perpendicular to the average groundwater flow direction, respectively) can be set by the user to match the measured dimensions of the study site, or for conceptual analyses, can be set to correspond to a square the size of an acre (208.7 ft * 208.7 ft).
- $L_{INFILTRATION}$ – The infiltration rate is best derived from a site-specific analysis of recharge. One method of determining the infiltration rate involves dividing the Model 1 value established for R by the estimated number of days required to accumulate that recharge amount. For example, if a 90-day water table fluctuation analysis was used to determine a value of R of 1.89 feet, the average rate of infiltration can be estimated as: 1.89 ft/90 days = 0.021 ft/day. If an annual rate of recharge is used to determine a daily infiltration rate for a scenario, users should adjust the value for seasonal variability as necessary.
- b – The average saturated thickness of the mixing zone is not necessarily equal to the full saturated thickness of the unconfined aquifer. In many shallow aquifers, particularly those comprised of stratified deposits, dissolved-phase groundwater contaminant plumes originating from the infiltration of vadose-zone leachate often remain located within the upper portions of the saturated zone (Domenico and Schwartz, 1998; Spitz and Moreno,

[4] For example, the $C_{LEACHATE\ NO3}$ concentration prediction from NO3-LEACHATE-Scenario 3 will be carried forward for use in Scenario 3 of the GWNO3-FORECAST model.

1996). Ideally, the mixing zone thickness is determined from a site-specific characterization of the vertical distribution of nitrate within the aquifer, as well as the site hydrogeology. The b variable also may be estimated through calibration.

- dH and K_H – The hydraulic properties of the saturated mixing zone are best determined from site-specific hydrogeologic investigations (e.g., slug testing, aquifer testing, water level measurements).
- $C_{GW\ INFLOW\ NO3}$ – This value is best determined from site-specific monitoring of conditions immediately upgradient of the site of interest. If a site measurement of upgradient nitrate-N concentration is unavailable, a default concentration for groundwater unaffected by human activities of 2 mg NO_3-N/L may be used (Tesoriero and Voss, 1997; Ryker and Frans, 2000; USGS, 1999; Madison and Brunett, 1985). The model user should, however, understand that many areas of Washington State have already been impacted by anthropogenic loading of nitrogen to groundwater. This suggests that current ambient background concentrations of nitrate-N at many sites in Washington may be elevated above what would occur naturally.

  Users interested in determining the portion of the $C_{GW\ OUTFLOW\ NO3}$ concentration that is specifically attributed to the leachate contribution can run a scenario with the $C_{GW\ INFLOW\ NO3}$ term set to a value of zero (0). This removes from consideration any nitrate mass inputs from upgradient groundwater sources and highlights the effect of the site leachate on groundwater quality.
- AP_{SZ} – This value is also best determined from direct field evidence (see for example Carey and Harrison, 2014). If a site-specific value for this variable is unavailable, a default value of 0% is recommended for the initial model run (i.e., conservative transport). If the $C_{GW\ OUTFLOW\ NO3}$ concentrations predicted by the GWNO3-FORECAST model are not in sufficiently close agreement with those measured in groundwater, the AP_{SZ} value may be determined through model calibration. Previous research has revealed that the denitrification capacity of aquifer systems may be limited and can be depleted over time, particularly in settings with high rates of surface irrigation and nitrogen application (Green and Bekins, 2010; Green et al., 2008).
- <u>Field measurements</u> of $C_{GW\ OUTFLOW\ NO3}$ (for model calibration) – To calibrate the GWNO3-FORECAST model prediction to the nitrate concentration measured in groundwater, the user should account for the model assumptions when selecting the location(s) and screen interval position of the well(s) acting as the data source for this value.

  Because the model assumes that the ambient groundwater and the leachate are fully mixed, the $C_{GW\ OUTFLOW\ NO3}$ model predictions best represent an average steady-state condition, not necessarily the condition at any particular point. As a result, it is probably best to calibrate the model to a concentration that represents multiple well locations. Wells located in the downgradient portion of the site may better reflect the assumption of complete mixing than wells located near the upgradient model boundary. Data should be drawn from wells that are constructed with open intervals positioned at depths that are representative of the mixing zone thickness (b) selected for modeling.

Groundwater velocities and plume transport times may need to be considered when selecting field measurements of groundwater nitrate-N for the purpose of calibrating the model for a specific scenario. Although the model provides a steady-state concentration prediction, in the field there may be a time lag between the entry of the leachate into the aquifer and the arrival of the resulting leachate/groundwater mix at the selected observation well(s).

GWNO3-FORECAST Model Limitations

The GWNO3-FORECAST model greatly simplifies the process of a chemical solution entering and mixing with groundwater in an underlying aquifer. The model assumptions of uniformity and full mixing may not reflect real-world processes occurring in the subsurface. In reality, heterogeneity and anisotropy in the mixing zone aquifer material may result in incomplete mixing of the leachate with the inflowing groundwater (or non-uniform attenuation). In the field, this can lead to localized areas of the aquifer showing a significant impact from the leachate, while other areas may remain largely unaffected.

Similar to the NO3-LEACHATE model, GWNO3-FORECAST ignores the influence of time; the model simply assumes that the mixing of the leachate and ambient groundwater is complete by the end of the scenario period of interest. The model also ignores any delays in transport time of the vadose-zone leachate to the water table; that process is assumed to be complete by the beginning of the GWNO3-FORECAST scenario.

Users should remember that the GWNO3-FORECAST model predictions represent an average steady-state condition for the entire mixing zone volume. As a result, the user should take care to avoid unrealistic model scenarios or input values that significantly violate the model assumptions described above.

Model 3 - GWNO3-BACKCAST

GWNO3-BACKCAST Model Description

The four-step GWNO3-BACKCAST model is similar to Models 1 and 2, but the models are combined and equations modified to back-calculate the leachate concentration, nitrate mass load, and average shallow soil nitrate concentration values necessary to maintain a known or desired groundwater nitrate-N outflow concentration (Figure 4).

For the purpose of predicting the key model variables, the GWNO3-BACKCAST model differs from the NO3-LEACHATE model in two important ways: (1) the $NO3_{SUPP}$ term is not used in the model equations, and (2) soil nitrate concentration averages are back-calculated assuming that all nitrate mass derived directly from the vadose zone is distributed uniformly throughout the top 2 feet of the soil column.

In Step 1, a modified version of the mass balance equation (Equation 9, Model 2) is used to back-calculate the leachate concentration $\left(\overleftarrow{\boldsymbol{C}}_{LEACHATE\ NO3}\right)$ that will produce a given groundwater nitrate-N outflow concentration ($\overleftarrow{\boldsymbol{C}}_{GW\ OUTFLOW\ NO3}$). The equation accounts for user-defined saturated zone attenuation processes (e.g., denitrification), aquifer conditions, and mixing.

Step 2 uses a modified version of Equation 8 from Model 1 to back-calculate the mass of leachable nitrate-N $\left(\overleftarrow{NO3}_{TOT\ LEACHABLE}\right)$ required to produce the $\overleftarrow{\boldsymbol{C}}_{LEACHATE\ NO3}$ value predicted by Step 1. User-defined leachate infiltration rate and vadose-zone attenuation effects are accounted for as necessary.

In Step 3, the nitrate-N mass contributed to the leachate by the recharge ($NO3_R$) is subtracted from the $\overleftarrow{NO3}_{TOT\ LEACHABLE}$ value, using a modified version of Equation 4. The result represents the nitrate mass that originates directly from within the vadose zone. Assuming that this mass is uniformly distributed within the top 2 feet of the soil column, the result is assigned the variable name: $\overleftarrow{NO3}_{LEACHABLE\ 0-2}$.

Finally, in Step 4, the model back-calculates the average 0-2 foot soil nitrate-N concentration ($\overleftarrow{\boldsymbol{C}}_{SOIL\ NO3\ 0-2}$) from the $\overleftarrow{NO3}_{LEACHABLE\ 0-2}$ mass estimate (factoring in a user-defined average soil bulk density value for that portion of the soil column). This is the average soil nitrate-N concentration in the top 2 feet of the soil column that will maintain a given $\overleftarrow{\boldsymbol{C}}_{GW\ OUTFLOW\ NO3}$ value, under the stated assumptions.

Model 3 can be run for ten independent model scenarios. Model predictions are automatically graphed by the spreadsheet file to allow visual evaluation of the results (Figure 5).

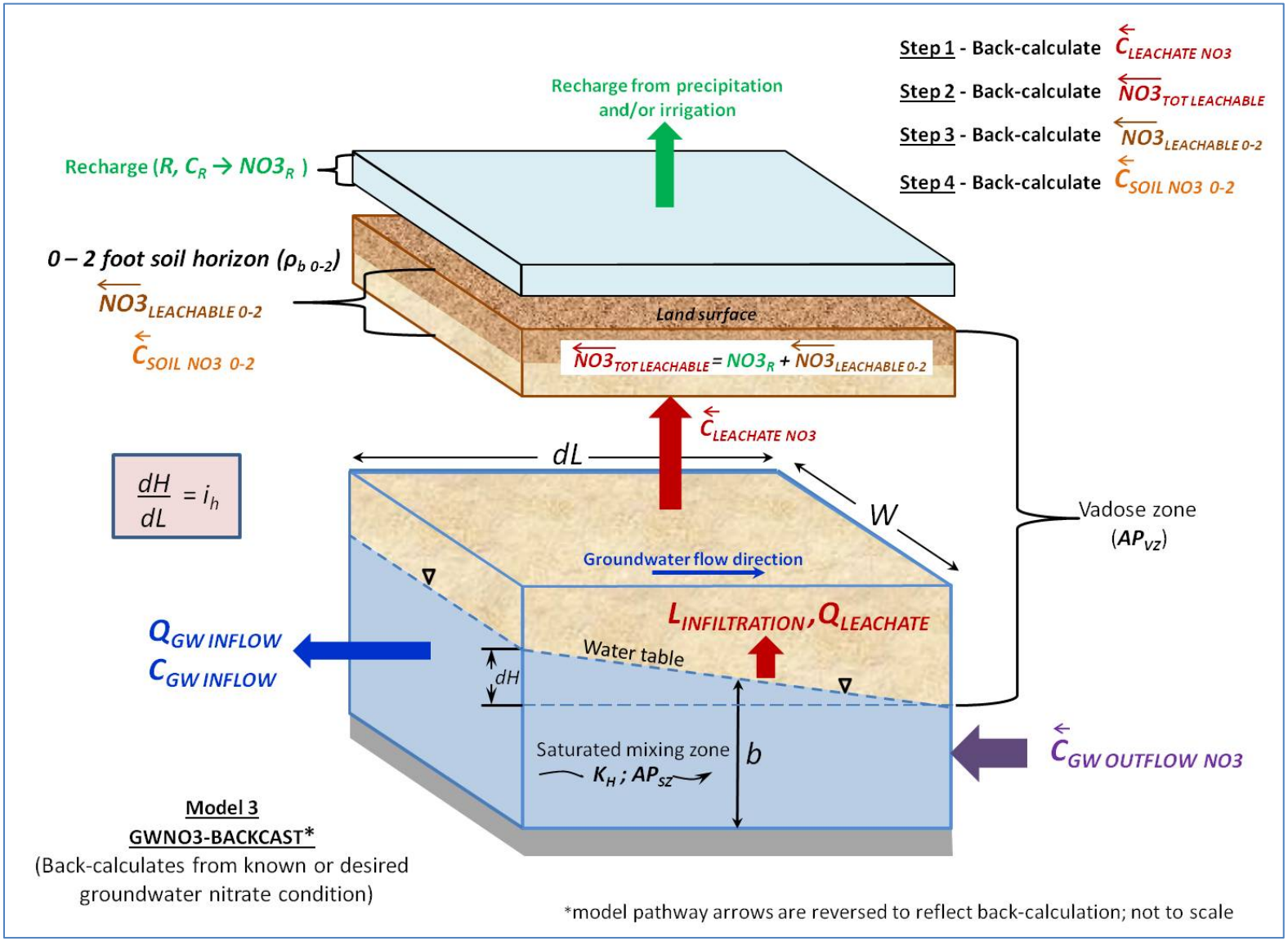

Figure 4. Conceptual Model Schematic – Model 3 (GWNO3-BACKCAST).
Refer to text for further explanation of figure symbols.

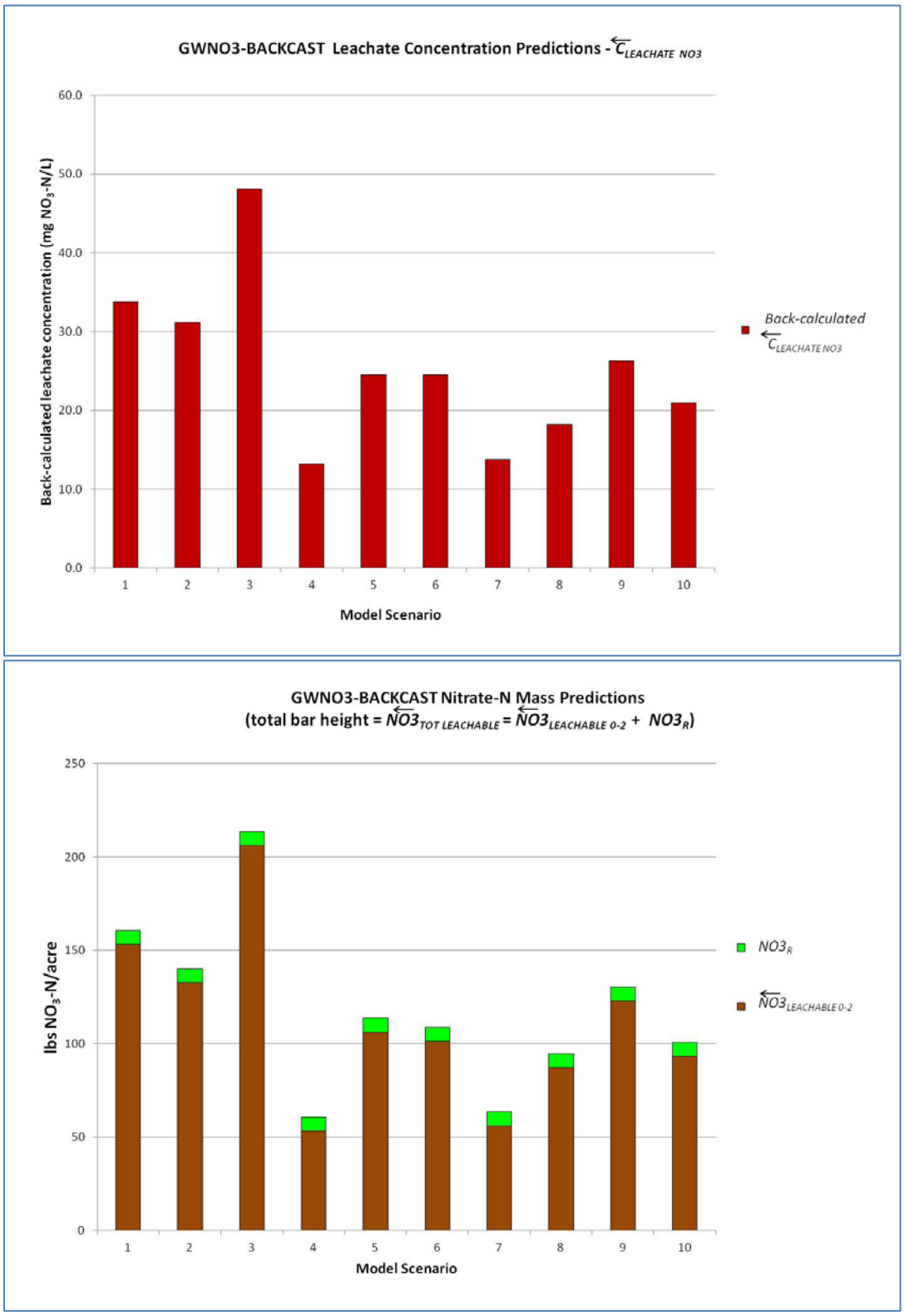

Figure 5. Example output graphs from the GWNO3-BACKCAST model spreadsheet.

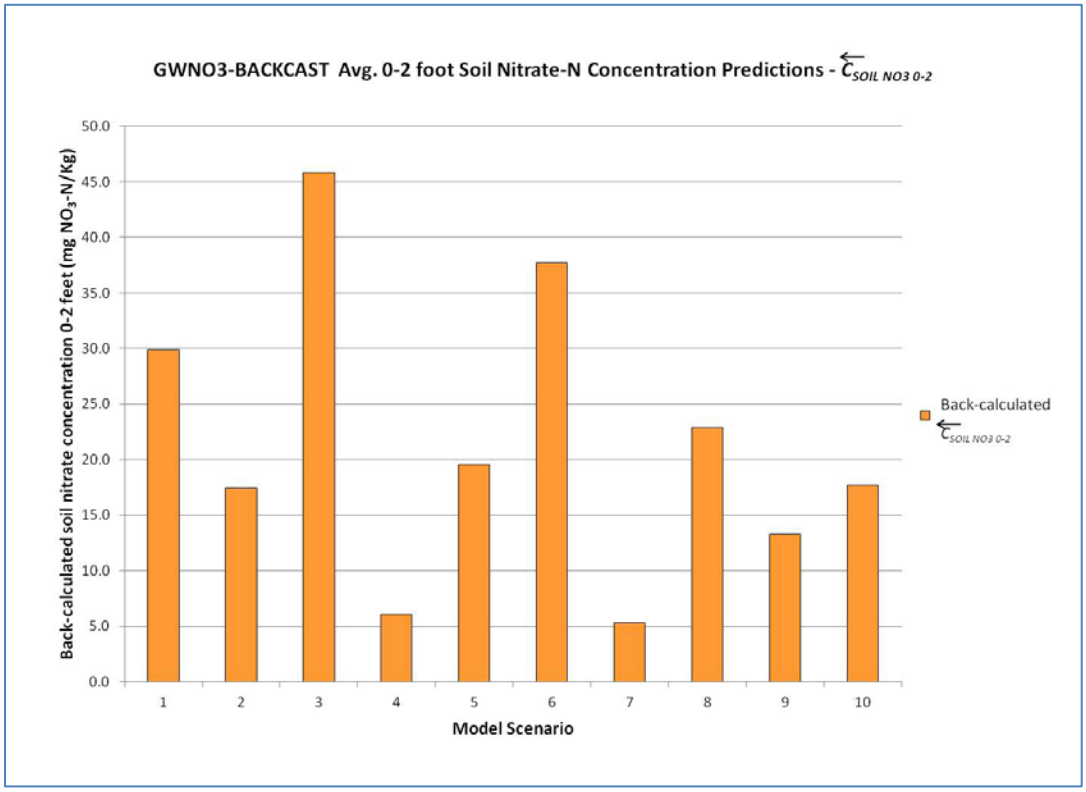

Figure 5 (cont.). Example output graphs from the GWNO3-BACKCAST model spreadsheet.

GWNO3-BACKCAST Model Equations

Step 1

To back-calculate the leachate concentration that will produce a given groundwater nitrate outflow concentration (under the stated assumptions), Equation 9 is modified to:

$$\overleftarrow{C}_{LEACHATE\ NO3} = \frac{\overleftarrow{C}_{GW\ OUTFLOW\ NO3}(Q_{LEACHATE} + Q_{GW\ INFLOW}) - \left[Q_{GW\ INFLOW} * C_{GW\ INFLOW\ NO3} * \left(1 - \frac{AP_{SZ}}{100}\right)\right]}{Q_{LEACHATE}\left(1 - \frac{AP_{SZ}}{100}\right)}$$

(Eq. 13)

where, for each model scenario:

$\overleftarrow{C}_{LEACHATE\ NO3}$ = back-calculated concentration of nitrate-N in leachate that will produce a given $\overleftarrow{C}_{GW\ OUTFLOW\ NO3}$ value (mg NO_3-N/L).

$\overleftarrow{C}_{GW\ OUTFLOW\ NO3}$ = <u>known or desired</u> nitrate-N concentration in groundwater exiting the downgradient boundary of the saturated mixing zone at the end of the scenario period of interest (mg NO_3-N/L).

Step 2

The amount of leachable nitrate-N mass required to produce the predicted $\overleftarrow{C}_{LEACHATE\ NO3}$ value from Step 1 is back-calculated by modifying Equation 8:

$$\overleftarrow{NO3}_{TOT\ LEACHABLE} = \frac{2.719\left(R * \overleftarrow{C}_{LEACHATE\ NO3}\right)}{\left(1 - \frac{AP_{VZ}}{100}\right)}$$

Eq. 14

where:

$\overleftarrow{NO3}_{TOT\ LEACHABLE}$ = back-calculated leachable nitrate-N mass required to produce the $\overleftarrow{C}_{LEACHATE\ NO3}$ value from Step 1 (lbs NO_3-N/acre).

Step 3

The nitrate-N mass contributed to the leachate directly from within the vadose zone can then be back-calculated using a modified version of Equation 4:

$$\overleftarrow{NO3}_{LEACHABLE\ 0-2} = \overleftarrow{NO3}_{TOT\ LEACHABLE} - NO3_R$$

Eq. 15

where:

$\overleftarrow{NO3}_{LEACHABLE\ 0-2}$ = back-calculated nitrate-N mass contributed to the leachate from the vadose-zone soils (lbs NO_3-N/acre). *Assumes all nitrate mass directly originating from the vadose zone is uniformly distributed within the top 2 feet of the soil column.*

Step 4

Assuming that the Step 3 $NO3_{LEACHABLE\ 0\text{-}2}$ mass is uniformly distributed within the top two feet of the soil column:

$$\overleftarrow{C}_{SOIL\ NO3\ 0-2} = \left(\frac{\overleftarrow{NO3}_{LEACHABLE\ 0-2}}{5.438\rho_{b\ 0-2}}\right) \qquad \text{Eq. 16}$$

where:

$\overleftarrow{C}_{SOIL\ NO3\ 0-2}$ = back-calculated average concentration of nitrate-N in the top 2 feet of the soil column required to produce a given $\overleftarrow{C}_{GW\ OUTFLOW\ NO3}$ value (mg NO_3-N/Kg DW).

$\rho_{b\ 0\text{-}2}$ = average soil bulk density for the 0-2 foot soil horizon (g/cm^3).

See Appendix A for the derivation of the conversion factors used in the above equations.

GWNO3-BACKCAST Model Assumptions

Most of the assumptions presented for Models 1 and 2 also apply to the GWNO3-BACKCAST model.

GWNO3-BACKCAST Model User-Defined Variables

The following variables can be defined by the user in the GWNO3-BACKCAST model:

- $\overleftarrow{C}_{GW\ OUTFLOW\ NO3}$ – Model 3 can be used to develop predictions about the nitrate concentration and mass conditions necessary to maintain groundwater at or below a desired concentration (e.g., the state groundwater quality criterion of 10 mg NO_3-N/L). Alternatively, it can be used to estimate the conditions that are required to explain a known groundwater concentration (e.g.: What nitrate concentration and mass loading conditions were necessary to produce the average groundwater nitrate concentration of 36 mg NO_3-N/L I observed near the downgradient boundary of my study field?).
- $C_{GW\ INFLOW\ NO3}$ – see notes for this variable in the Model 2 section ("GWNO3-FORECAST User-Defined Variables").
- AP_{SZ} – see notes for this variable in the Model 2 section.
- dL and W – see notes for these variables in the Model 2 section.
- $L_{INFILTRATION}$ – see notes for this variable in the Model 2 section.
- b (used in Eq. 11 to compute $Q_{GW\ INFLOW}$) – To determine the amount of nitrate-N loading that will result in a given concentration of nitrate in groundwater *at the water table*, the user should set this value to zero (0). Doing so also sets $Q_{GW\ INFLOW}$ to zero, removing from consideration mixing and dilution within the saturated zone, and resulting in the assumption that $\overleftarrow{C}_{LEACHATE\ NO3} = \overleftarrow{C}_{GW\ OUTFLOW\ NO3}$ (assuming AP_{VZ} and AP_{SZ} are set to 0%).

To determine the loading required to produce a given $\overleftarrow{C}_{GW\ OUTFLOW\ NO3}$ value after mixing within the aquifer has occurred, the user should set the value of b to reflect the assumed vertical extent of mixing. See additional notes for this variable in the Model 2 section.

- K_H *and* dH (used in Eqs. 11 and 12 to compute $Q_{GW\ INFLOW}$) – These values are best determined from site-specific investigations.
- R – see notes for this variable in the Model 1 section ("NO3-LEACHATE Model User-Defined Variables").
- AP_{VZ} – see notes for this variable in the Model 1 section.
- $C_{R\ NO3}$ – see notes for this variable in the Model 1 section.
- $\rho_{b\ 0\text{-}2}$ – see notes for estimation of soil bulk density in the Model 1 section and in Appendix B. This value should represent the average bulk density for the top 2 feet of the soil column.

GWNO3-BACKCAST Model Limitations

All of the limitations described for Models 1 and 2 also apply to GWNO3-BACKCAST. The assumption that the leachable nitrate mass directly derived from the vadose zone is entirely concentrated within the top 2 feet of the soil column ignores the possibility that some of that mass could actually originate from below 2 feet. The assumption that the $NO3_{LEACHABLE\ 0\text{-}2}$ nitrate mass is *uniformly* distributed within the 0-2 foot soil horizon results in a prediction of the *average* soil nitrate concentration that would produce the given $\overleftarrow{C}_{GW\ OUTFLOW\ NO3}$ value. In the field, soil nitrate is often not uniformly distributed with depth.

Spreadsheet Models

The three models described above have been incorporated into an interactive Excel 2007® file. Instructions for using the models are embedded in the spreadsheet file. Model predictions are automatically graphed and, if appropriate, can be compared to field-collected data for calibration purposes.

It is recommended that users thoroughly review the information in this publication, with a particular eye toward understanding the model assumptions and limitations, before applying the models to site-specific loading analyses.

Conclusions and Recommendations

- The three models described in this report represent a significant simplification of the real world. The different assumptions used by the model can lead to either under-predictions or over-predictions of actual (i.e., measured) concentration and mass values, depending on how well those assumptions align with existing conditions *in situ*. These models are therefore best used for rapid, low-cost screening analyses, range-finding exercises, and scenario comparison (e.g.: What will happen if we reduce the nitrate-N concentration of the irrigation water we apply? Is site A more vulnerable to groundwater contamination than site B? What sort of nitrate mass loading is required to explain the existing groundwater nitrate condition we see at our site?). The more the user can rely on site-specific field measurements to inform the input values, and the closer the tested scenarios adhere to the model assumptions, the less uncertainty there will be in the model predictions.

- Calibration of the GWNO3-FORECAST predictions to nitrate concentrations measured in groundwater can strengthen user confidence in those predictions and may help reveal processes that are perhaps otherwise difficult to quantify in the field. For example, some users may observe insufficient agreement between model predictions and field measurements unless they increase the input values for supplemental nitrate mass in the vadose zone. This suggests the possibility that previously unrecognized mineralization of organic nitrogen may be occurring at their site, or that other sources of nitrate have not been accounted for. This approach may also reveal denitrification that had not been previously detected. Users are reminded that model solutions are not necessarily unique (i.e., there may be multiple combinations of input variable values that can produce the same model prediction).

- The models also can be a useful tool for sensitivity analyses, to determine which input parameters have the greatest or least influence on the prediction results. Sensitivity analyses can be a cost-effective way to identify priorities for more in-depth field investigation. For example, if the model predictions are found to be highly sensitive to the amount of recharge, additional field measurements and quantitative analysis of recharge may be justified to increase confidence in the model predictions. Varying a single parameter over multiple scenarios, while fixing all other input parameters at appropriate constant values, is a simple way to conduct a sensitivity analysis. For advanced users interested in using automated tools to run multi-parameter sensitivity and uncertainty analyses, we recommend tools such as the Department of Ecology's freeware YASAIw add-on for Excel (see: www.ecy.wa.gov/programs/eap/models.html), or proprietary software such as Oracle's Crystal Ball.

- The models are designed to predict spatially averaged, steady-state conditions; they are not intended to predict nitrate loading or concentrations at any particular point in the subsurface. Several simplifying assumptions, such as complete mixing, limit the usefulness of the models for very short periods of time; the models are best suited for longer-term analyses.

- Because the models described in this report significantly simplify the processes controlling the fate and transport of nitrate in both the vadose zone and the saturated zone, the model predictions should not be used as a replacement for direct monitoring of groundwater conditions.

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Appendices

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Appendix A. Derivation and Units of Conversion Factors

A.1 Soil concentration-to-mass unit conversion

To convert a 1-foot thick soil sample nitrate-N concentration in units of mg/Kg to a nitrate-N mass value in units of lbs/acre, first:

$$\frac{1\ kg}{1000000\ mg} * \frac{1000\ mg}{1\ g} * \frac{2.205*10^{-6}\ lbs}{1\ mg} * \frac{1.233*10^{9}\ cm^3}{1\ acre\ ft} = 2.719\ \frac{kg\ lbs\ cm^3}{g\ mg\ acre\ ft} \qquad \text{Eq. A.1}$$

Combining the units above with the units for soil bulk density and soil volume yields:

$$\frac{kg\ lbs\ cm^3}{g\ mg\ acre\ ft} * \frac{g}{cm^3} * \frac{1\ acre\ ft}{acre} = \frac{kg\ lbs}{mg\ acre} \qquad \text{Eq. A.2}$$

Then, multiplying the Eq. A.2 units by the given soil nitrate-N concentration units:

$$\frac{kg\ lbs}{mg\ acre} * \frac{mg}{kg} = \frac{lbs}{acre} \qquad \text{Eq. A.3}$$

In the GWNO3-BACKCAST model, the conversion factor of 2.719 is multiplied by 2 (2.719*2 =5.438) to help convert a nitrate-N soil mass back to an average 0-2 foot soil nitrate-N concentration.

A.2 Dissolved phase concentration-to-mass unit conversion

To convert a dissolved phase nitrate concentration in recharge, in units of mg/L, to a mass value in units of lbs/acre, first:

$$\frac{2.205*10^{-6}\ lbs}{1\ mg} * \frac{1.233*10^{6}\ L}{1\ acre\ ft} = 2.719\ \frac{lbs\ L}{mg\ acre\ ft} \qquad \text{Eq. A.4}$$

Combining the conversion factor units with the units for recharge volume and recharge concentration yields:

$$\frac{lbs\ L}{mg\ acre\ ft} * \frac{acre\ ft}{acre} * \frac{mg}{L} = \frac{lbs}{acre} \qquad \text{Eq. A.5}$$

A.3 Volume-to-volume unit conversion

$$\frac{28.32\ L}{ft^3} * \frac{ft^3}{day} = \frac{L}{day} \qquad \text{Eq. A.6}$$

Appendix B. Soil Bulk Density

This appendix presents analyses of bulk density value ranges for cropland soils in eastern and western Washington. The analyses were conducted and reported by Steve Campbell, Soil Scientist with the U.S. Dept. of Agriculture, Natural Resources Conservation Service (NRCS), West National Technology Support Center. Data for the analyses were drawn from the NRCS Soil Survey Geographic Database (SSURGO). Bulk density value ranges are presented for the 0-30 cm (~0-12") and the 30-60cm (~12-24") depth-intervals.

Spreadsheets of the raw SSURGO data compiled for the analysis (by soil map unit) are available as supplemental files to this report (to support site-specific bulk density determinations).

Soil Bulk Densities in Eastern Washington Soils used for Cropland - June 2014

SSURGO soil survey bulk density data for the 0-12 inch (0-30 cm) and 12-24 inch (30-60 cm) depths were evaluated for soils used for cropland in Major Land Resource Areas 7, 8, and 9 in Eastern Washington.
In general, sandy soils have the highest bulk density, loamy soils are intermediate, and clayey soils have the lowest bulk density (www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_053256.pdf).
The table below from the Soil Survey Manual, Chapter 3, contains groupings of soil texture classes (www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/ref/?cid=nrcs142p2_054253).

General Terms[a]	Texture Classes
Sandy soil materials:	
Coarse	Sands (coarse sand, sand, fine sand, very fine sand) Loamy sands (loamy coarse sand, loamy sand, loamy fine sand, loamy very fine sand)
Loamy soil materials:	
Moderately coarse	Coarse sandy loam, sandy loam, fine sandy loam
Medium	Very fine sandy loam, loam, silt loam, silt
Moderately fine	Clay loam, sandy clay loam, silty clay loam
Clayey soils:	
Fine	Sandy clay, silty clay, clay

Range of Eastern Washington Soil Bulk Densities for Cropland Soils (grams per cubic centimeter).

Texture Class Group	Count	Low bulk density	Average bulk density	High bulk density
Coarse, 0-12 in.	154	1.30	1.50	1.60
Coarse, 12-24 in.	188	1.35	1.55	1.65
Moderately coarse, 0-12 in.	547	1.20	1.30	1.50
Moderately coarse, 12-24 in.	371	1.25	1.35	1.55
Medium, 0-12 in.	1761	1.15	1.25	1.45
Medium, 12-24 in	1652	1.20	1.30	1.50
Moderately fine, 0-12 in.	79	1.15	1.25	1.45
Moderately fine, 12-24 in.	258	1.20	1.30	1.50
Fine, 0-12 in.	14	1.10	1.20	1.35
Fine, 12-24 in.	86	1.15	1.30	1.50
Organic, 0-12 in.	1	0.20	0.20	0.25
Organic, 12-24 in.	1	0.20	0.20	0.25

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Phone: 503-273-2421
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Soil Bulk Densities in Western Washington Soils used for Cropland - June 2014

SSURGO soil survey bulk density data for the 0-12 inch (0-30 cm) and 12-24 inch (30-60 cm) depths were evaluated for soils used for cropland in Major Land Resource Area 2 in Western Washington.
In general, sandy soils have the highest bulk density, loamy soils are intermediate, and clayey soils have the lowest bulk density (www.nrcs.usda.gov/Internet/FSE_DOCUMENTS/nrcs142p2_053256.pdf).
The table below from the Soil Survey Manual, Chapter 3, contains groupings of soil texture classes (www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/ref/?cid=nrcs142p2_054253).

General Terms[a]	Texture Classes
Sandy soil materials:	
Coarse	Sands (coarse sand, sand, fine sand, very fine sand) Loamy sands (loamy coarse sand, loamy sand, loamy fine sand, loamy very fine sand)
Loamy soil materials:	
Moderately coarse	Coarse sandy loam, sandy loam, fine sandy loam
Medium	Very fine sandy loam, loam, silt loam, silt
Moderately fine	Clay loam, sandy clay loam, silty clay loam
Clayey soils:	
Fine	Sandy clay, silty clay, clay

Range of Western Washington Soil Bulk Densities for Cropland Soils (grams per cubic centimeter).

Texture Class Group	Count	Low bulk density	Average bulk density	High bulk density
Coarse, 0-12 in.	125	1.00	1.40	1.65
Coarse, 12-24 in.	236	1.05	1.45	1.65
Moderately coarse, 0-12 in.	342	0.75	1.15	1.55
Moderately coarse, 12-24 in.	312	0.75	1.20	1.85
Medium, 0-12 in.	597	0.70	1.10	1.50
Medium, 12-24 in	395	0.75	1.15	1.55
Moderately fine, 0-12 in.	98	0.75	1.20	1.40
Moderately fine, 12-24 in.	163	0.90	1.25	1.45
Fine, 0-12 in.	8	0.95	1.20	1.45
Fine, 12-24 in.	68	1.20	1.30	1.50
Organic, 0-12 in.	69	0.20	0.25	0.50
Organic, 12-24 in.	65	0.20	0.25	0.5

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Appendix C. Estimated Background Concentrations of Dissolved Nitrate-N Concentrations in Washington State Precipitation

To provide the basis for a default assumption for the background or initial concentration of nitrate (nitrate-N or NO_3-N) in rainfall-derived recharge, precipitation-weighted annual mean concentration data from the National Atmospheric Deposition Program (NAPD) website (http://nadp.sws.uiuc.edu/) were compiled from seven National Trends Network (NTN) stations (see Table A-1). Data from five of these stations were used to calculate a composite average value for western Washington; data from the other two stations were used to calculate a composite average value for eastern Washington.

Table A-1. Precipitation-weighted annual mean nitrate-N concentration data from selected Washington State monitoring stations (source: NAPD).

NADP NTN Station ID	County, State	Latitude	Longitude	Data Period	Average NO_3-N Concentration (mg/L)	95% Confidence Interval (mg/L)
Western Washington						
WA14	Jefferson, WA	47.8597	-123.933	2002-2012	0.022	±0.002
WA19	Skagit, WA	48.5403	-121.446	2002-2012	0.059	±0.004
WA21	Pierce, WA	46.8353	-122.287	2002-2012	0.065	±0.006
WA98	Skamania, WA	45.5694	-122.21	2002-2012	0.064	±0.008
WA99	Pierce, WA	46.7582	-122.124	2002-2012	0.042	±0.004
Composite average NO_3-N concentration (mg/L)* : 0.05						
Eastern Washington						
WA24	Whitman, WA	46.7606	-117.185	2003-2012	0.086	±0.007
ID02	Bonner, ID	48.3518	-116.84	2003-2012	0.088	±0.009
Composite average NO_3-N concentration (mg/L)*: 0.09						

*NTN data are reported as NO_3 in mg/L. These values were converted to NO_3-N using a conversion factor of 0.2259.

Appendix D. Units of Measure and Conversion Factors

Units of Measure

acre-ft	acre-foot
L	liters
L/day	liters per day
mg NO_3-N/Kg	milligrams nitrate as nitrogen per kilogram = parts per million
ft	feet
lbs NO_3-N/acre	pounds nitrate as nitrogen per acre
lbs N/acre	pounds nitrogen per acre
mg NO_3-N/L	milligrams nitrate as nitrogen/liter
g/cm^3	grams per cubic centimeter
ml/g	milliliters per gram

Appendix E. Supplemental Microsoft Excel 2007 Files

Three Microsoft Office Excel 2007® worksheet files are included with this report:

- ***NO3_MB_loading_models_V1.0.xlsx***
  This worksheet file contains the three spreadsheet models described in this report (NO3-LEACHATE, GWNO3-FORECAST, and GWNO3-BACKCAST).

- ***USDA_Campbell_W_WA_soil_BD_data.xlsx***
  This worksheet file contains soil bulk density data for crop soils in western Washington, drawn from the NRCS SSURGO database (Campbell, 2014).

- ***USDA_Campbell_E_WA_soil_BD_data.xlsx***
  This worksheet file contains soil bulk density data for crop soils in eastern Washington, drawn from the NRCS SSURGO database (Campbell, 2014).

These Excel files are available only on the Internet, linked to this report at https://fortress.wa.gov/ecy/publications/SummaryPages/1403018.html

Nitrogen Dynamics at a Manured Grass Field Overlying the Sumas-Blaine Aquifer in Whatcom County

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- Appendices B through T
- Response to Comments

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Cover photo: Liquid manure application by subsurface deposition (photo by Barbara Carey).

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Nitrogen Dynamics at a Manured Grass Field Overlying the Sumas-Blaine Aquifer in Whatcom County

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Water Resource Inventory Area (WRIA) and 8-digit Hydrologic Unit Code (HUC) numbers for the study area:

- WRIA: 1 - Nooksack
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Abstract

The Washington State Department of Ecology, in cooperation with Washington State University, conducted a 4-1/2-year intensive monitoring study at a manured grass field overlying the Sumas-Blaine Aquifer in northwest Washington. The purpose of the study was to evaluate nitrogen dynamics in dairy manure, soil, crop, and groundwater at a field overlying the Sumas-Blaine Aquifer.

We quantified the mass of nitrogen added to the field in the form of manure, inorganic fertilizer, and irrigation water; the mass of nitrogen removed in the crop; the mass of nitrate in the soil during the post-harvest period; and the concentration of nitrate in groundwater beneath the field. Shallow depth to water (0 to 11 feet) enabled rapid responses to nitrate transport, especially during the high-rainfall period (October through March).

Average monthly nitrate concentrations in 6 shallow monitoring wells ranged from 5.5 to 30 mg/L-N with a maximum in one well of 45 mg/L-N. Early winter average nitrate concentrations in groundwater, representing newly recharged water carrying nitrate from the soil column, were (1) above 10 mg/L-N following growing seasons with nitrogen loading greater than the mass of nitrogen removed in the crop and (2) generally below 10 mg/L-N when nitrogen loading was similar to crop removal. Other factors that affected nitrate concentrations in groundwater included timing of manure applications, tillage of the field shortly before the study began, and denitrification in the aquifer.

Model results based on measured field parameters indicated an average of 115 lb/acre of nitrate leached to groundwater from September through March. Two methods for estimating the nitrogen residual at the end of the growing season, mass balance analysis and post-harvest soil nitrate testing, were not reliable predictors of nitrate concentrations in groundwater. Direct monitoring of water quality at the water table was the only accurate and reliable method for tracking effects of manure management on groundwater nitrate.

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Executive Summary

Government, community, and agricultural groups around Washington State have been working to determine the main causes of nitrate contamination in groundwater and to identify cost-effective ways to address the problem. One of the dominant sources of nitrate loading to groundwater in the state, and around the United States, is nitrogen releases from agricultural practices.

The northwestern portion of Whatcom County is an area of high-intensity agricultural production. The main agricultural businesses in Whatcom County are dairy farming and berry production. Conventional practice for both types of operation is applying nitrogen-bearing fertilizer. Whatcom County has the second highest number of dairy cows in the state and the highest intensity of raspberry cultivation in the country.

Groundwater supply in this area is derived almost exclusively from the Sumas-Blaine Aquifer (SBA), an unconfined aquifer occurring in the unconsolidated glacial deposits that blanket the region (Figure ES-1). Over the last 30 years, this area has had one of the highest percentages of water supply wells in the state failing to meet the drinking water standard for nitrate (29% of wells tested had concentrations greater than 10 mg/L as nitrogen). Groundwater is the only source of drinking water for residents living in the northern, rural part of the county. As of 2010, the population living over the SBA not on city water systems was 18,000 to 27,000 people.

Factors that make groundwater in Whatcom County particularly sensitive to water quality impacts from intensive agricultural production include:

- Shallow depth to water.
- Relatively permeable character of the aquifer deposits.
- Long period of heavy rainfall each year.

Combined with the high mobility of nitrate in the environment, these characteristics enable rapid transport of nitrate from surface soils to the water table.

The Washington State Dairy Waste Nutrient Management Act[1] of 1998 requires that all dairies have approved Dairy Nutrient Management Plans (DNMPs) in place by 2003. The primary objective of the law is to ensure that surface water and groundwater quality in the state are not adversely affected by dairy manure.

The study described in this Executive Summary is one of a series of assessments the Washington State Department of Ecology (Ecology) has conducted over several decades to characterize the extent and nature of groundwater nitrate conditions in the SBA. We hope that insights gained from this study will guide stakeholders and decision-makers in efforts to restore and protect local and state groundwater resources.

[1] Chapter 90.64 Revised Code of Washington (RCW)

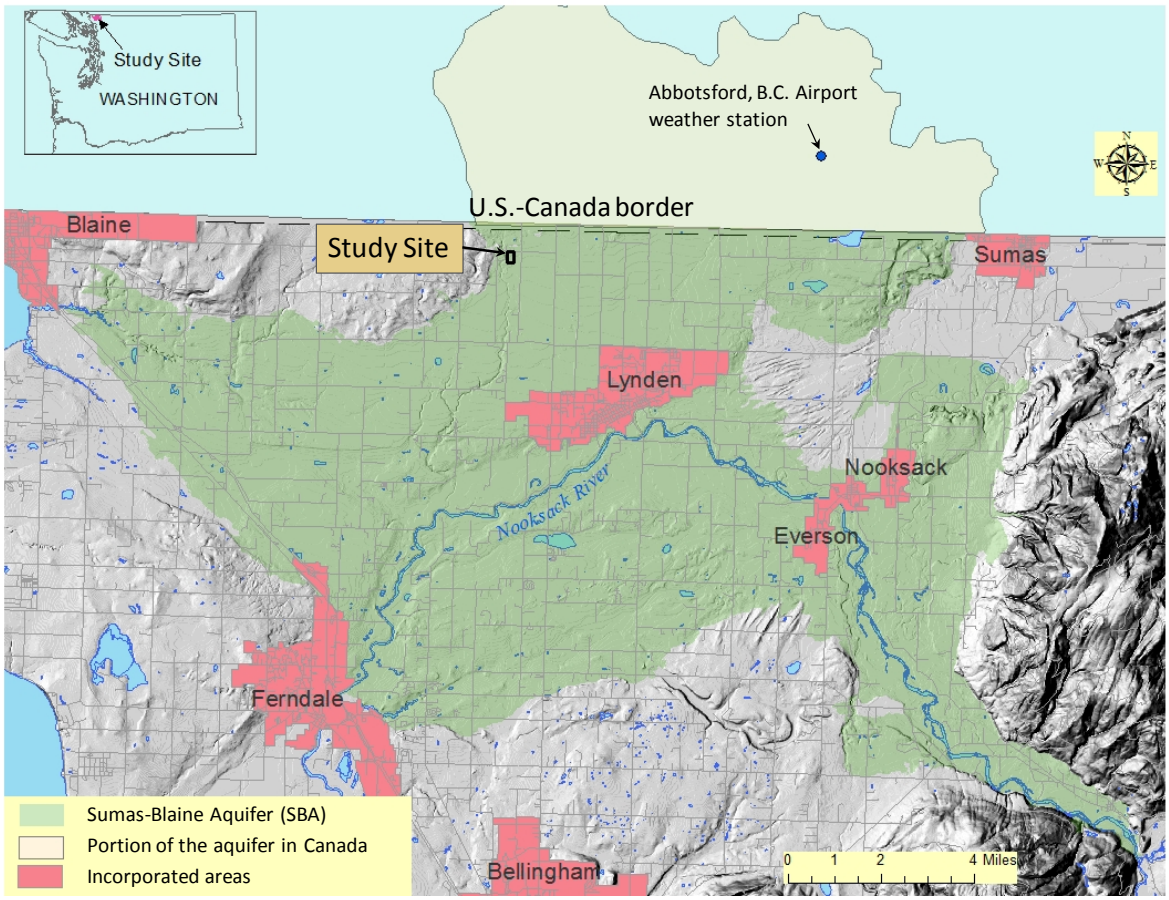

Figure ES-1. Sumas-Blaine Aquifer and study site location.

Purpose and objectives of the study

In 2003, Ecology's Bellingham Field Office requested that Ecology's Environmental Assessment Program conduct a field study to evaluate the effectiveness of DNMPs in protecting the quality of the SBA. One of the measures required in DNMPs is that producers collect samples of soil nitrate at the end of the growing season to assess nutrient balance. These soil samples were intended to provide a general assessment of nutrient balance at a field, not for assessing groundwater protection.

The main purposes of the study were to:

- Improve our understanding of the nitrogen dynamics and fate at a manured field.
- Evaluate the effectiveness of land application guidelines of DNMPs in protecting the quality of the SBA at the study site.

The technical objectives of the study were to:

- Track nitrogen dynamics and fate at a grass field overlying the SBA that receives manure applications:
  - Quantify the sources of nitrogen input and output to the study field on an annual basis.

  - Quantify changes in soil nitrate and groundwater quality conditions at the study field over time.
  - Compare the soil and groundwater results to Washington State guidelines and standards.
- Identify the key farm management and environmental factors that influence groundwater and soil nitrate conditions.
- Perform an annual nitrogen mass balance evaluation for the study field.
- Use the nitrogen mass balance and soil nitrate sampling results to estimate the annual nitrogen residual mass available for leaching to groundwater.
- Evaluate the correlation of the nitrogen residual estimates to groundwater results, and determine reliability of these estimates for predicting groundwater quality responses to manure management practices.
- Evaluate whether current guidelines for manure management and soil monitoring are adequately protective of groundwater.
- Evaluate study findings in the context of the SBA as a whole.

Background

The study site is a 22-acre grass field located in northwestern Whatcom County, Washington, about 3 miles north of the town of Lynden and 0.3 mile south of the Canada border (Figure ES-1). The site lies on the flat Lynden Terrace, a glacial outwash plain that slopes gradually southward to the Nooksack River.

The regional climate is humid maritime with mild temperatures and high wintertime precipitation due to proximity to the Pacific Ocean. Annual local precipitation ranges from 32 inches in the southwest part of the SBA to over 60 inches in Abbotsford, British Columbia, Canada. Roughly 60 to 70% of the annual precipitation typically occurs from October through March, outside of the typical growing season for crops.

Hydrogeology and soils

The SBA is the uppermost hydrogeologic unit in the area, covering about 150 square miles. The aquifer consists of unconsolidated sand and gravel outwash with minor clay lenses. The aquifer averages 50 feet in thickness. The depth to water is less than 10 feet over all but a small portion of the aquifer in the east, making the aquifer highly susceptible to surface contamination. The underlying hydrogeologic units are not feasible sources for large-scale drinking water consumption. Recharge to the SBA occurs mainly from precipitation that occurs from September through March.

The study site, located on the western edge of the SBA, is an area dominated by finer-grained material at the surface compared to other aquifer locations. The depth to the bottom of the aquifer is 40 feet at the site based on well borings from the site. Hale silt loam soil overlies the study site. Hale soils are part of the Lynden-Hale-Tromp grouping that overlies much of the SBA.

Dairy field management

The 22-acre study field has received manure for over 20 years at about the same application rate (~400 to 700 lb total N/acre/year) as during the study, according to the dairy producer. The site was planted in grass before the study and was tilled and re-seeded back to grass in April 2004 using conventional tillage practice. During the study, the dairy producer managed the field as before the study. Manure was mostly applied using an aerator (also referred to as subsurface deposition).

Manure was typically applied 3 to 5 times per year following each grass cutting. The final manure application for any given study year occurred between the end of August and early October. In 2 of the 4 study years (2005 and 2006), the final manure application occurred after the last crop harvest.

Each summer, irrigation water from a nearby shallow well was applied at the study site. The grass crop was harvested 4 to 5 times each year.

Study design

Ecology's Environmental Assessment Program partnered with the Washington State University Livestock Nutrient Management Program (WSU) to design an intensive, multi-media, multi-year monitoring study at a grass field overlying the SBA that received applications of manure. Participants from WSU focused on monitoring and characterizing the manure, soil, and crop, while Ecology focused on monitoring groundwater conditions underlying the study field.

We calculated the nitrogen mass balance of the study field each year and compared the estimated residual nitrogen to the shallow underlying groundwater nitrate concentrations. We also evaluated the effect of various environmental and management factors on the nitrogen mass balance and groundwater nitrate concentrations.

Our monitoring program focused on the following components of the nitrogen cycle to evaluate the balance of nitrogen at the study field system as shown in Figure ES-2:

- Inputs
  - Manure and inorganic fertilizer (mass of nitrogen applied to the field)
  - Irrigation water (volume and nitrogen concentration added to the field)
- Outputs
  - Grass harvested (mass of nitrogen removed from the field)
- Residual
  - Soil (fall nitrate mass in the top 1-foot of soil)
  - Groundwater near the top of the water table (nitrogen concentration)

To support the evaluation of nitrogen transport at the study site, additional field work was conducted to characterize the hydrogeology and soil characteristics of the study site. This included:

- Measuring static water levels in monitoring wells.
- Testing the hydraulic characteristics of the aquifer underlying the study field.
- Conducting grain size analysis of site deposits.
- Measuring chloride in groundwater to use as a conservative (non-reactive) tracer.
- Measuring other water quality constituents in groundwater that contribute to understanding nitrate levels.

An important factor that influenced the study results, although not part of the study design, is that the dairy producer tilled the grass field and replanted it in grass in April 2004, 4 months before the study began. Tillage typically increases nitrate mineralization and is often followed by increased nitrate leaching.

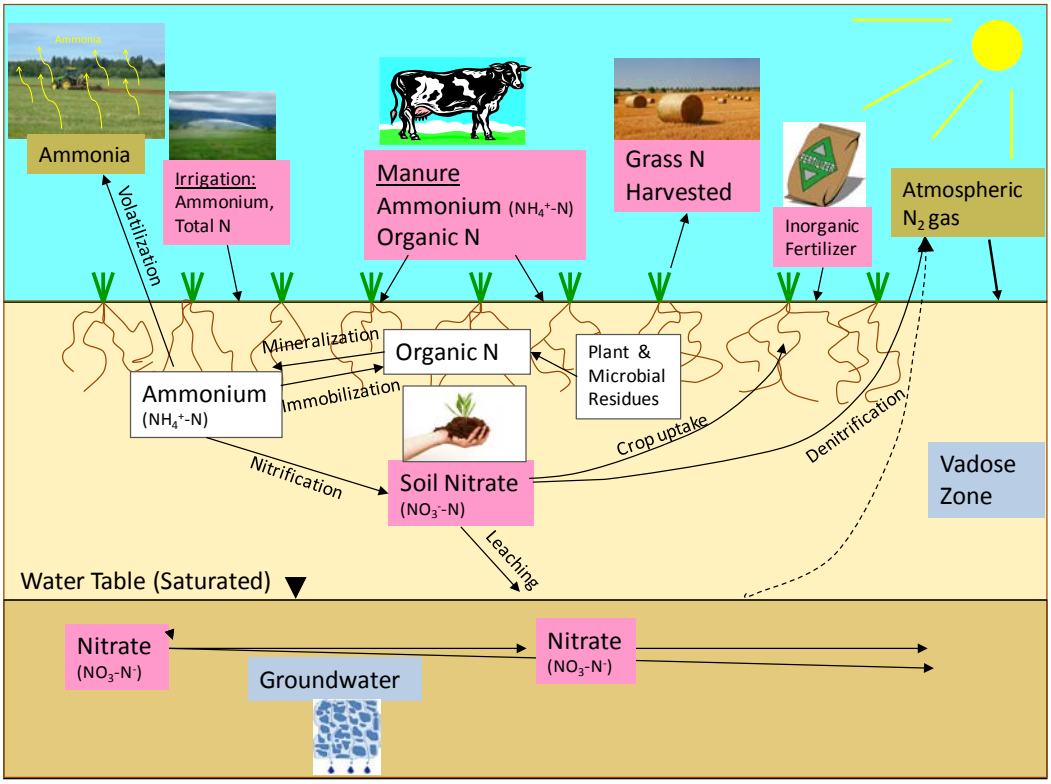

Figure ES-2. Major nitrogen components in the study field.
Media in pink boxes were monitored. Items in brown boxes were estimated.

Major findings

Groundwater sampling results

Groundwater sampled near the top of the water table represented recently recharged groundwater. Nitrate concentrations in individual monitoring wells varied widely on most dates (Figure ES-3). Fifty-six percent of the average monthly groundwater nitrate values over the course of the study were above 10 mg/L-N.

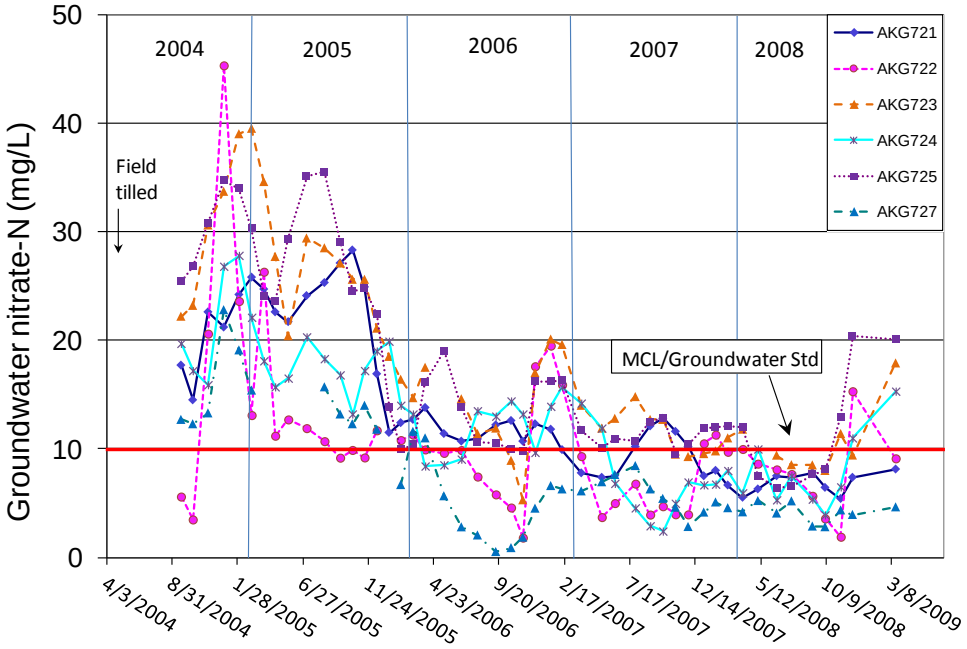

Figure ES-3. Shallow groundwater nitrate-N concentrations in individual monitoring wells.

Nitrate concentrations did not meet the groundwater quality/drinking water standard at the beginning of the study in 2004. The field had recently been tilled and replanted, which contributed to higher nitrate loading to groundwater than normal. However, ratios of groundwater nitrate-N (nitrate as nitrogen) to chloride indicated the main cause for the initial high concentrations was nitrogen loading from manure application. Groundwater nitrate gradually declined through the summer of 2008 due to lower manure application. At the end of the growing season in 2008, nitrate concentrations began to increase again, rising above the drinking water standard in 4 of the 6 monitoring wells.

Influencing factors

Several environmental and management factors influenced nitrate concentrations in soil and groundwater during the study.

Environmental factors

Hydrogeologic conditions

The depth to water at the site ranged from 0 to 11 feet below ground surface with the highest water table conditions occurring during the wet non-growing season (October through March). This allowed for rapid transport of available nitrate to groundwater.

Fine-grained material at the site likely influenced the substantial nitrogen loss observed in 4 of the 6 monitoring wells via microbially-mediated denitrification, which is less likely in coarse-grained materials.

Soil conditions

Higher moisture content in soils during the wet season generally led to higher permeability and rapid nitrate leaching. Summertime drying, which we observed in the beginning of the study, can slow mineralization and diminish crop uptake. This in turn can increase the amount of residual nitrate available to leach to groundwater in the fall. When irrigation water was applied earlier in the 2007 growing season, subsequent groundwater nitrate concentrations were lower than similar years without irrigation.

Heavy rainfall in the fall apparently led to a sharp increase in soil nitrogen mineralization at a time when crop growth was slowing. Most of the newly mineralized nitrate apparently leached to groundwater.

Precipitation

Most of the annual precipitation in the area occurred during a period of limited crop growth (October through March). Precipitation that infiltrated through the soil to the groundwater (recharge) carried available nitrate to the water table.

Upgradient conditions

Groundwater at the water table represents the most recently recharged water. We constructed all but 1 of the 7 monitoring wells with open intervals intersecting the water table. This allowed us to track groundwater responses to nitrate entering the aquifer in the near vicinity of the well. An analysis of groundwater travel times indicated that samples collected from the shallow monitoring wells represented recently recharged water that likely entered the aquifer within the study field boundaries. Based on this and other lines of evidence, we concluded that upgradient influences on groundwater quality results were probably limited.

Management factors

Rate of nitrogen application (External loading)

The more nitrogen applied to the field in excess of the crop demand, the higher the amount of nitrate that reached the water table in our study. Higher shallow groundwater nitrate concentrations occurred in the early winter following years with nitrogen application in excess of crop uptake (2005, 2008); concentrations were lower when the amount of nitrogen applied was less than the crop uptake (2006, 2007) (Figure ES-4).

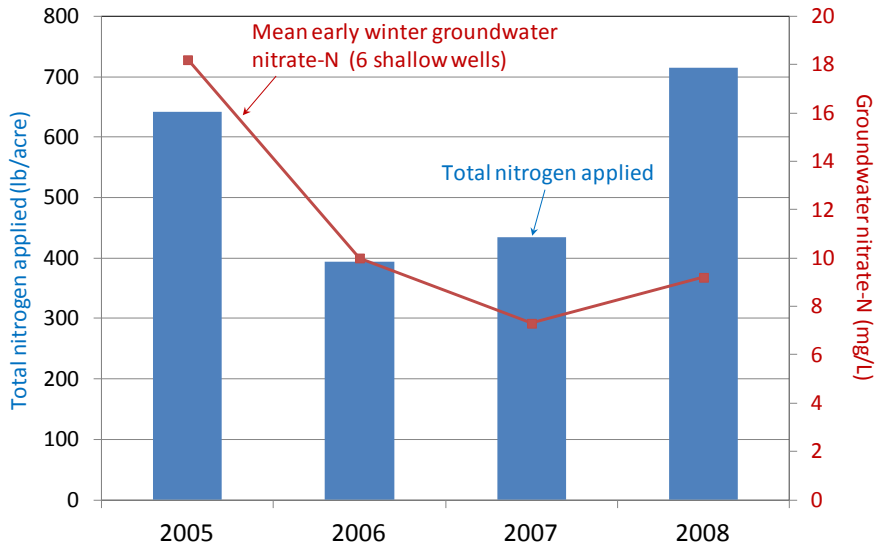

Figure ES-4. Total nitrogen mass applied annually (external loading) from manure and inorganic fertilizer (bars) and the mean early winter (November-December) groundwater nitrate-N concentration in 6 shallow wells (line).

Rate of nitrogen mineralization (Internal loading)

Organic nitrogen in the soil from previous years and manure organic nitrogen from the current year are bacterially converted each year to plant-available nitrogen. Accurately quantifying the amount and timing of this internal loading is difficult. The annual average estimated amount of mineralized nitrogen was roughly one-half of the annual mass of plant-available nitrogen (Figure ES-5).

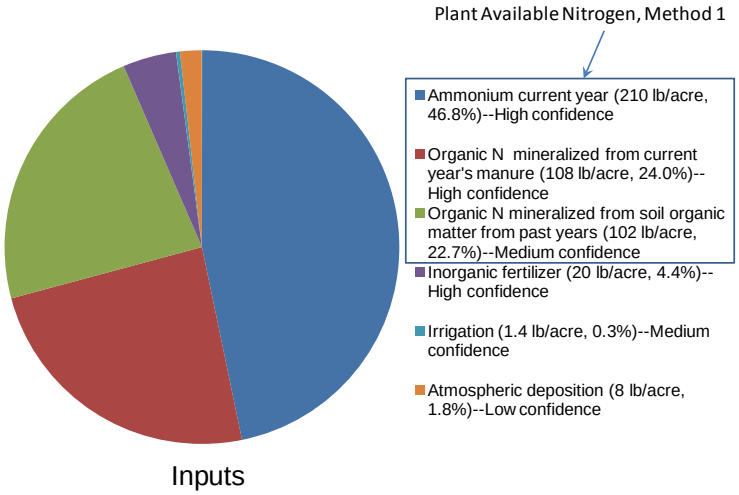

Figure ES-5. Average annual nitrogen inputs to the soil column during the study in lb/acre and our relative degree of confidence in the numbers.

Mineralization of organic nitrogen contributed to groundwater loading throughout the year, including the high-precipitation, non-growing season.

Tillage effects

Tillage of the study field in April 2004, 4 months before the start of the study, contributed to nitrate loading to groundwater the first year of the study but was not the dominant cause of high groundwater nitrate concentrations the first year.

Crop removal

Annual nitrogen removal in the crop was fairly consistent from year to year (393 to 457 lb/acre) despite large variations in the nitrogen application rate (394 to 715 lb/acre). This indicates that nitrogen was probably not the limiting factor for crop growth.

Timing of manure application

Applying manure too late in the growing season resulted in nitrate increases in groundwater. Similarly, applying manure too early in the season was followed by increased groundwater nitrate concentrations.

Denitrification

Denitrification in shallow groundwater at the site was controlled by reduced dissolved oxygen (DO) concentrations in 4 of the 6 wells. We estimated that an average of 28% of the nitrate in the 4 low-DO wells was converted to nitrogen gas and lost to the atmosphere. Denitrification was more prevalent in the summer and fall, when little oxygenated recharge water reached the water table.

Annual nitrogen residual estimates – correlation with groundwater nitrate

An important goal of the DNMPs is to minimize the amount of residual nitrogen left in the field after the growing season. This minimizes the amount of nitrate available to leach to groundwater. We evaluated 2 field-based methods of estimating the amount of residual nitrogen left in the study field at the end of each growing season:

1. Mass balance estimate of annual nitrogen residual
2. Fall soil nitrate residual (1 foot depth)

The main input variable for the mass balance analysis is the amount of plant-available nitrogen (PAN). We used the detailed nitrogen application data collected during the study with the most current method for estimating PAN for western Washington and Oregon.

Using the mass balance analysis, the annual nitrogen residual (N_{Excess}) from 2005 to 2008 was -79 to 146 lb/acre with a 4-year average of 19 lb/acre (Figure ES-6). Significant nitrogen deficits were predicted for 2006 and 2007. The annual mass balance residual did not correspond well with the nitrogen application rates or with groundwater concentrations. The main source of uncertainty in the mass balance evaluation was the contribution from mineralized organic matter from past years.

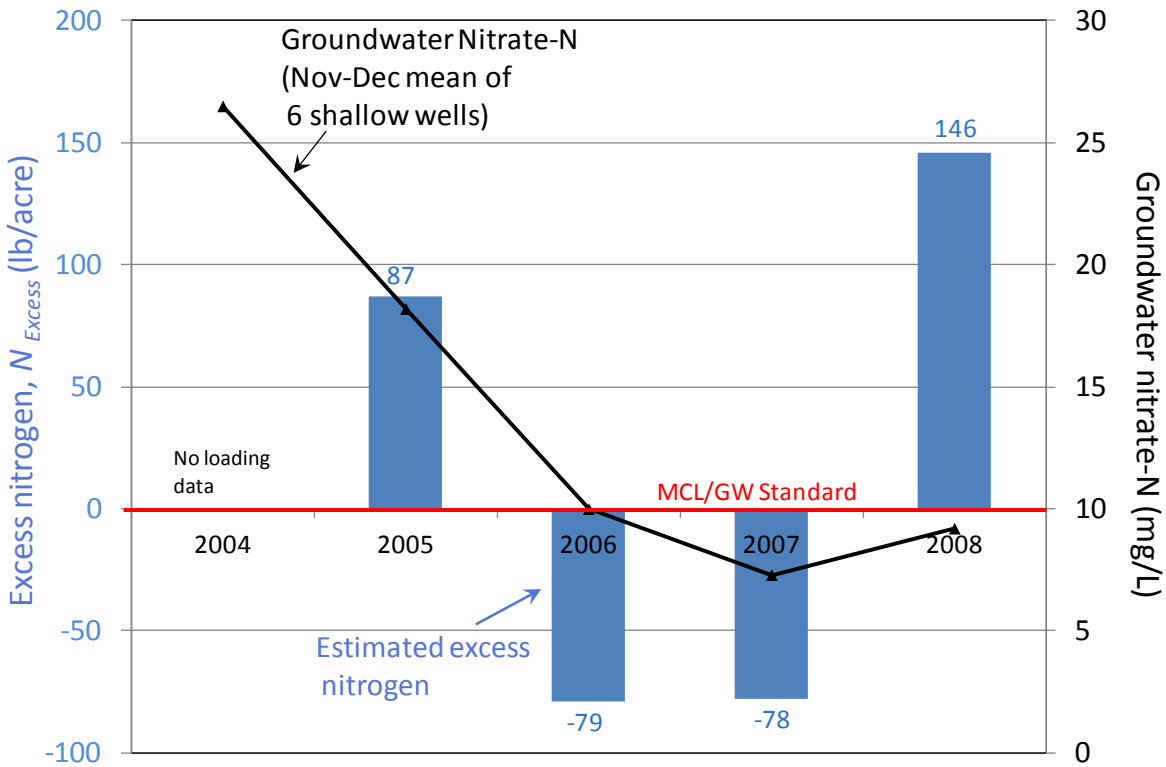

Figure ES-6. Estimated excess nitrogen N_{Excess} (bars) calculated using mass balance vs. annual mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

r^2=0.22
MCL = Maximum contamination level
GW = Groundwater

The second method used the fall soil nitrate values collected weekly in September and October to estimate the annual mean and maximum soil nitrate residual. Soil nitrate concentrations varied greatly over time as shown in Figure ES-7. The 4-year average fall (September-October) soil nitrate result was 72 lb/acre; the maximum was 104 lb/acre. These averages were 4 to 5 times the mass balance average and were highly variable over short time periods. Soil nitrate residual values did not track closely with nitrogen application rates.

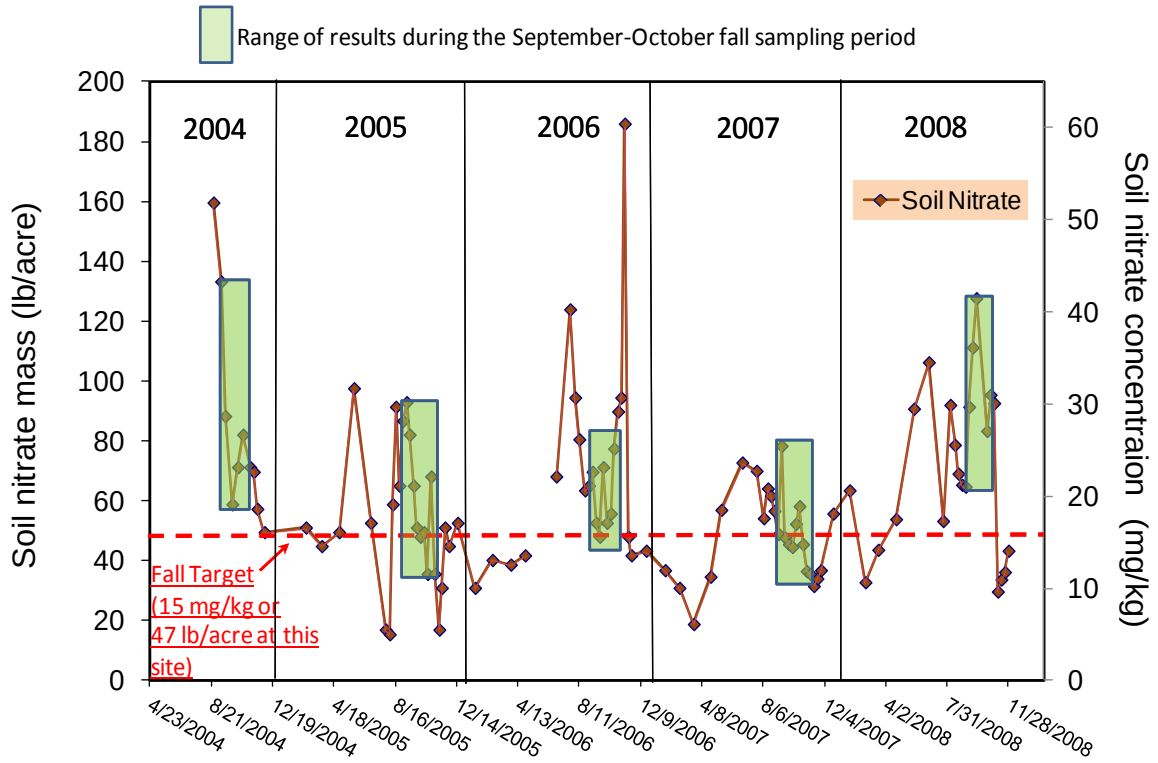

Figure ES-7. Soil nitrate results at 1-foot depth.

Green shaded areas indicate results for the typical fall soil sampling time, September through October. Red dashed line indicates the level below which management changes are not recommended based on Sullivan and Cogger (2003).

Fall soil nitrate results are typically an underestimate of the amount of nitrate available for leaching at the end of the growing season, because any nitrate already leached cannot be measured. In addition, nitrate deeper in the soil, as well as nitrate generated in the soil after October 31, are not included in fall soil nitrate results.

We attempted to correlate results of the nitrogen mass balance method and the fall soil nitrate method with early winter groundwater nitrate concentrations, because early winter is typically the time when the end-of-season residual nitrate in the soil reaches the water table. Neither approach correlated well with groundwater nitrate concentrations, as shown in Figures ES-6 and ES-8.

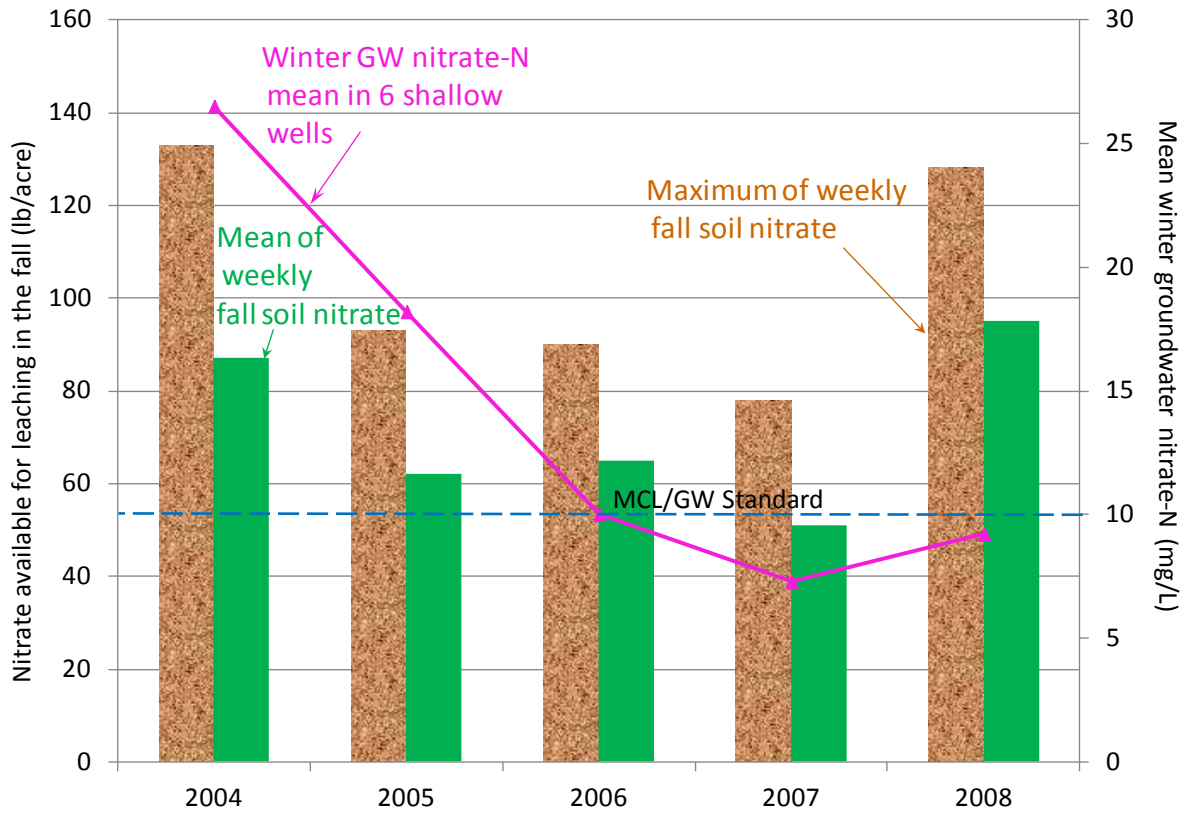

Figure ES-8. Comparison of fall soil nitrate mean concentrations (green untextured bars) vs. maximum concentrations (brown textured bars) with mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

r^2=0.30 for maximum fall soil nitrate.
r^2=0.07 for mean fall soil nitrate.

Because mass balance and soil nitrate methods were not good predictors of nitrate leaching, we used an alternative method for estimating residual nitrate. The GWNO3-BACKCAST model was used to predict the amount of nitrate necessary to produce the groundwater nitrate concentrations observed. The model is based mainly on measurements at the study site (recharge rates, groundwater nitrate concentrations, hydraulic conductivity, and horizontal gradients).

On average, the BACKCAST model back-calculated an annual nitrate loading to groundwater of 115 lb/acre during the wet season, with a range of 42 to 230 lb/acre (Figure ES-9). Seasonal back-calculated averages were 66 lb/acre during the late fall/early winter and 49 lb/acre during the late winter/early spring. The BACKCAST model predictions were most sensitive to the rate of recharge, the denitrification rate in groundwater, and the thickness of the groundwater mixing zone.

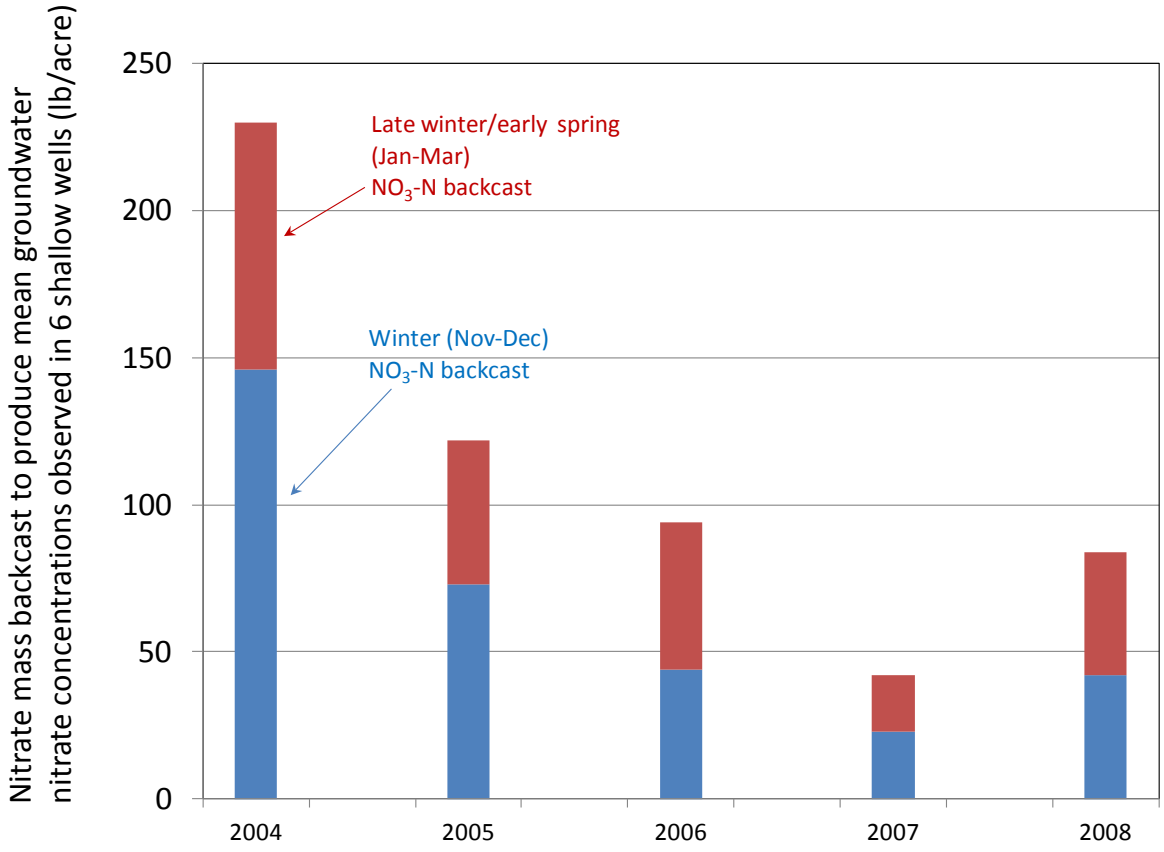

Figure ES-9. Total wet-season nitrate mass load predicted using the GWNO3-BACKCAST model.

Soil nitrate as an indicator of leaching to groundwater

We evaluated the target post-harvest soil nitrate concentration recommended for dairies in western Washington, 15 mg/kg, in relation to the recharge amounts observed. The calculated leachate nitrate concentration that would result from combining the fall soil nitrate threshold concentration for grass (15 mg/kg) with the observed annual volumes of recharge ranged from 11 to 18 mg/L-N. This does not include nitrate available for leaching below the top foot. All soil nitrate samples collected according to the protocols for post-harvest soil nitrate testing during the study exceeded the 15 mg/kg target.

If the seasonal *average* September to mid-November soil nitrate concentrations was used in the same calculations with the recharge that occurred during the study, the estimated nitrate concentration in leachate would have been 16 to 23 mg/L-N. If the seasonal *maximum* September to mid-November soil nitrate concentration was used in the calculation, the estimated leachate nitrate concentration would have been 29 to 50 mg/L-N.

Recommendations

Reducing nitrate leaching to groundwater at manured dairy fields, like the one in this study over the Sumas-Blaine Aquifer (SBA), will require improving manure application based on evolving science and technology. This includes nitrogen loading analyses that take groundwater into account. Groundwater monitoring will be needed to evaluate the effectiveness of measures to reduce nitrate loading to groundwater.

Based on the results of this study, the following actions are recommended to promote improvements to groundwater quality in the SBA and in other areas of Washington State with similar conditions.

Reduce nitrate loading to groundwater

- Develop a process whereby (1) manure and fertilizer nitrogen inputs and outputs are tracked on a field-by-field basis and (2) the information is used to minimize nitrate leaching below the root zone. Because of high seasonal rainfall and shallow depth to water in the area, appropriate timing and amount of nutrient application is crucial. Involvement of state and local organizations in partnership with universities, dairy and other agricultural producers is needed to improve nitrogen use efficiency and protect groundwater quality.

- Review available mass balance assessment methods for use in Dairy Nutrient Management Plans (DNMPs). Develop or adapt existing methods so that they more accurately account for effects on groundwater (i.e., more accurate assessment of soil organic matter contribution).

- Calculate nitrogen applications (manure, fertilizer, irrigation water) and removal (crop removal) based on measured amounts and nitrogen analyses to estimate mass balance. This is especially important in areas where groundwater nitrate already does not meet the drinking water standard. Timing of nitrogen application relative to recharge is especially important.

- Our results indicate that it is best to schedule the last manure application by late August to early September. Manure application during the high-recharge period (September through mid-March) is likely to increase nitrate leaching to groundwater.
- Where groundwater is well-oxygenated and denitrification rates are low, take special care to prevent overapplication or application during the high-recharge period.
- If soil moisture is low in the summer, consider irrigating to increase mineralization and nitrification which may increase available nitrate for the crop. Avoid overapplication of irrigation water to prevent nitrate leaching.
- Consider extending the time between tillage events to decrease the amount of nitrogen reaching groundwater.
- Consider limiting manure application to forage crops during the first season following tillage.
- Consider updating the post-harvest soil nitrate test (PSNT) guidance to incorporate hydrologic influences (i.e., local aquifer recharge) based on the expertise of land grant universities as well as local and state scientists.
- Until revised, use the existing post-harvest soil nitrate protocols (methods and timing) and targets (15 mg/kg for grass; 20 mg/kg for corn) as criteria for evaluating manure management at dairy operations in western Washington.

Monitor to evaluate the effectiveness of management improvements

A program is needed to determine how well current and future manure management practices are working to improve groundwater quality. Because there is no reliable substitute, direct groundwater monitoring using dedicated monitoring wells is a key component of an effectiveness monitoring program.

Although groundwater monitoring is the only available way to determine the amount, or the concentration of, nitrate that actually reaches the water table, fall soil nitrate monitoring is a necessary tool for on-farm nutrient management. If conducted with limitations in mind, soil nitrate monitoring also can serve as a screening tool to focus closer inspection of groundwater conditions.

Nitrogen mass balance evaluations also are an important tool for dairy producers to manage nutrients and identify potential courses of action to address high soil and groundwater nitrate concentrations. Current methods for analyzing mass balances for DNMPs should be evaluated to ensure that the methods are as accurate as possible.

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Introduction

Background information

The Sumas-Blaine Aquifer (SBA) underlies about 150 square miles of U.S. land and is the primary source of drinking water for 18,000 to 27,000 residents of northwest Whatcom County, Washington (U.S. Census, 2010). The SBA is part of the larger Abbotsford-Sumas Aquifer that straddles the U.S.-Canada border. The aquifer averages 50 feet in thickness in the U.S. (Figure 1; Tooley and Erickson, 1996).

Groundwater within the SBA flows predominantly from north to south (British Columbia, Canada to the U.S.), but local patterns of flow are also affected by interactions with surface water features (Figure 1). The depth to water is less than 10 feet in most of the SBA but is more variable in British Columbia. In winter the depth to water is near the surface in much of the SBA, requiring artificial drainage to prevent flooding due to heavy precipitation in some places.

Nitrate in drinking water

Nitrate concentrations in excess of 10 mg/L-N, the maximum contaminant level (MCL) acceptable in public drinking water supplies[2] and the groundwater quality standards for Washington,[3] have been documented in the SBA for at least the past 23 years (Erickson and Norton, 1990; Garland and Erickson, 1994; Cox and Kahle, 1999; Erickson, 2000 and 1998; Carey, 2002; Almasri and Kaluarachchi, 2004; Mitchell et al., 2005; and Redding, 2008).

In 1997, 21% of 250 private wells tested in the SBA exceeded the drinking water limit (Erickson, 1998). In a 35-well subset of the 250 wells previously sampled, 71% contained nitrate at concentrations higher than 10 mg/L-N between 2003 to 2005 (Redding, 2008).

Several public water supply wells near the City of Lynden exceed the drinking water standard for nitrate, affecting over 1,000 residents (Pell, 2011).

Agricultural activities in the U.S. and British Columbia, Canada

Intensive agriculture has been conducted over the SBA for the past 50 years. Dairy farming has historically been the predominant agricultural activity over the aquifer, with raspberry and other berry production becoming more prominent in the past 20 years. There are approximately 37,000 acres in dairy production, 8,200 acres of raspberries, and 2,600 acres of blueberries in Whatcom County (Embertson, 2010; Whatcom Farm Friends, 2012).

Berry and poultry production have replaced most of the dairy land in the Abbotsford area of British Columbia. Zebarth et al. (1998) showed that much of the surplus nitrogen that leaches to groundwater or runs off to surface water on the Canadian side of the aquifer is due to changes in

[2] Chapter 246-290-310 Washington Administrative Code (WAC)
[3] Chapter 173-200 WAC

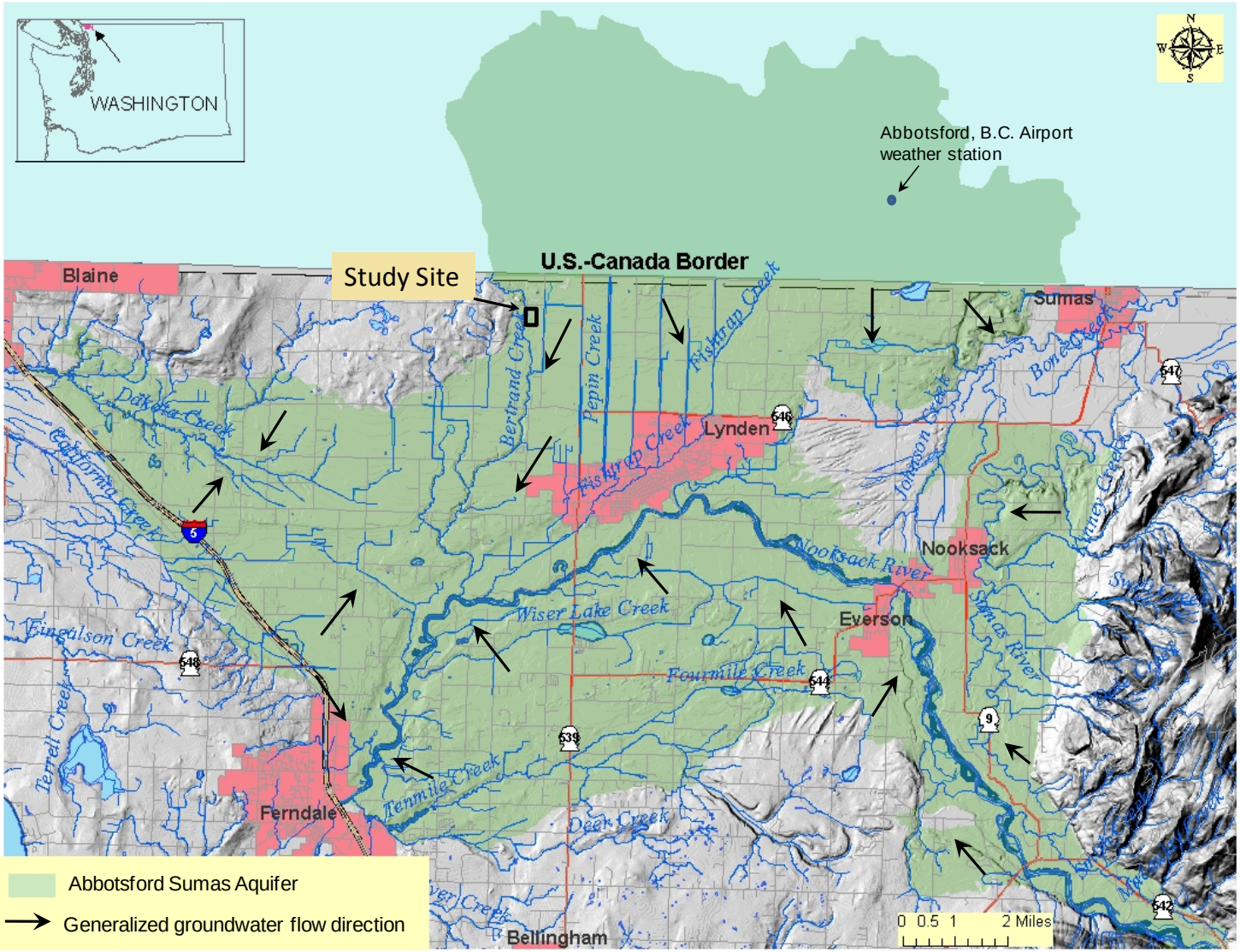

Figure 1. Study site location within the Abbotsford-Sumas Surficial Aquifer.
Groundwater flow direction arrows are from Erickson (1998) and Graham (2008).

agricultural practices over the past 40 years. Small fruit crops, which have replaced almost all of the cropland formerly in grass for dairy cows, take up only 10% of the amount of nitrogen taken up by forage crops, leaving more nitrogen available for infiltration below the root zone (Zebarth et al., 1998). Currently there are about 2,500 acres in raspberries and 1,000 acres in blueberries over the Canadian portion of the aquifer (Sweeney, 2012).

On the Washington side of the aquifer, the number of dairy farms dropped by one-half from roughly 2000 to 2010 (Embertson, 2010). Fields formerly planted in grass to feed dairy cows are being converted to crops that take up less nitrogen and, as a result, contribute a surplus of nitrogen to groundwater similar to that on the Canadian side of the aquifer.

Agricultural activities overlying the Canadian portion of the aquifer have also resulted in groundwater quality impacts (McArthur and Allen, 2005). The concentrations of nitrate along the Canada-U.S. border area are variable, with the highest concentrations on the eastern side of the aquifer.

Although the distribution of nitrate concentrations in groundwater entering the U.S. from the Canadian side of the aquifer system remains uncharacterized, groundwater typically flows horizontally in the direction of flow (generally north to south in the SBA) with solute concentrations dispersing deeper into the aquifer with distance from the source. Therefore, shallow groundwater in the U.S. would most likely not be affected by activities north of the border.

Adverse effects of high nitrate concentrations

High nitrate concentrations in drinking water can cause methemoglobinemia, or blue-baby syndrome, in infants. This potentially life-threatening condition is caused by conversion of nitrate to nitrite in the digestive system. The nitrite then reacts with iron in hemoglobin, restricting transport of oxygen to cells. An increased risk of spontaneous abortion or certain birth defects may be associated with drinking nitrate-contaminated water. Cancer risks also have been associated with elevated nitrate in water and food (Centers for Disease Control and Prevention, 1996; Chiu and Tsai, 2007; Ward et al., 2005; Weyer et al., 2001).

Besides human health effects of nitrate, nitrate in groundwater can adversely affect surface water by increasing primary productivity in streams, rivers, and lakes hydraulically connected to the aquifer system. When algal and plant material that depend on nitrogen decompose, oxygen depletion can adversely affect fish and other aquatic life (Matson et al., 1997; Howarth and Marino, 2006).

Application of manure to crops

Much effort has gone into developing nutrient management plans for dairies in the area, since the Washington Dairy Nutrient Management Act was adopted in 1998[4]. Yet questions remain about the best management practices necessary to simultaneously maintain crop health and reduce and prevent nitrate contamination in local groundwater. Because the SBA already displays a high vulnerability to nitrate leaching (Erwin and Tesoriero, 1997), and so much of the land overlying the aquifer receives dairy nutrients in the form of liquid manure, it is important to optimize nutrient management. Some of the issues of concern for land application of manure include:

- Rate of nitrogen application
- Timing of manure application
- Soil type (texture and organic matter influence)
- Methods for estimating plant-available nitrogen
- Methods for evaluating excess/deficit nitrogen (soil nitrate, leachate nitrate concentration, groundwater nitrate concentration)

While one goal of manure application is to apply an amount of nitrogen that will contribute to optimal crop growth, simultaneously achieving a close balance between inputs and outputs of nitrogen to protect groundwater quality is often elusive.

A number of studies have shown that measured concentrations of nitrate in soil or soil pore-water, or estimates of surplus nitrogen loading from mass balance surveys, are not reliable predictors of underlying groundwater nitrate concentrations (Viers et al., 2012; van der Schans et al., 2009; van Es et al., 2006; Basso et al., 2005; Zebarth et al., 1998; Bechmann et al., 1998). These methods can either overestimate or underestimate groundwater impacts. This is because transformations between various forms of inorganic and organic nitrogen are difficult to predict. However, Goss and Goorahoo (1995) found that although farm nitrogen budgets did not accurately predict groundwater nitrate concentrations, they were useful for identifying farms likely to cause environmental contamination.

The timing and amount of manure applied to crops have been found to be the key factors in maintaining nitrogen balance on manured fields (Oenema et al., 2010; Van Es et al., 2006; Verloop et al., 2006; Di and Cameron, 2002).

The amount and timing of precipitation that carries soil nitrate to the water table also plays a significant role in the concentration of nitrate ultimately reaching the water table (Sonneveld et al., 2010; Oenema et al., 2010; de Ruijter et al., 2007; Boumans et al., 2005; Zebarth, 1998). Smith et al. (2002) suggest that liquid manure applications not be made during wet winter months in "Nitrate Vulnerable Zones" of the United Kingdom, because heavy rainfall in winter months leaches soluble nitrate below the root zone.

[4] Chapter 90.64 Revised Code of Washington (RCW)

Nitrogen cycle

Nitrate is part of the dynamic system of nitrogen-containing compounds transformed in the environment and referred to as the *nitrogen cycle*. This section describes the major parts of the nitrogen cycle that occur at a typical manured field over the SBA.

Figure 2 shows the main components of the nitrogen cycle for the study site during the 2 main agricultural seasons.

- The top diagram in Figure 2 represents the spring/summer period, when the water table is several feet below the root zone of the grass crop. The vadose zone includes the root zone [roughly 0 to 3 feet below ground surface (BGS)] and extends to the water table, which during this drier season is roughly 10 feet BGS.
- The bottom diagram in Figure 2 shows the fall/winter scenario after heavy precipitation has raised the water table to within roughly 1 to 3 feet of the surface, frequently intersecting the root zone.

The first manure application to grass crops typically occurs in early spring, and additional applications are made after each cutting through the summer. The final yearly manure application usually occurs in September or October. The nitrogen composition of dairy manure varies depending on a number of variables including animal genetics, feeding programs, and available feed (ASAE, 2005).

Volatilization

A portion of the ammonium contained in manure converts to ammonia gas after application and volatilizes to the atmosphere. The amount that volatilizes depends on the application method, weather conditions (especially wind, rainfall, and temperature), and soil conditions. Most volatilization occurs during the drying process soon after manure is applied (Beegle et al., 2008; Sullivan, 2008). Therefore, rainfall or saturated soil conditions during or shortly after application can significantly limit volatilization. If ammonium infiltrates into the soil before drying, then less ammonia volatilizes than if drying had occurred. Chemical conditions in the soil also influence ammonia volatilization. Volatilization is higher where soil pH is high and cation exchange is low (Beegle et al., 2008).

Subsurface deposition, the principal method of manure application used at the study site, reduces the amount of ammonia volatilized to about 15% compared to 30% to 45% for spray application methods (Sullivan, 2008). Even with aerial spraying, ammonia volatilization decreases if manure is incorporated into the soil soon after application. Reduced ammonia volatilization using this method allows more of the nitrogen applied to be available for plant uptake or leaching to groundwater.

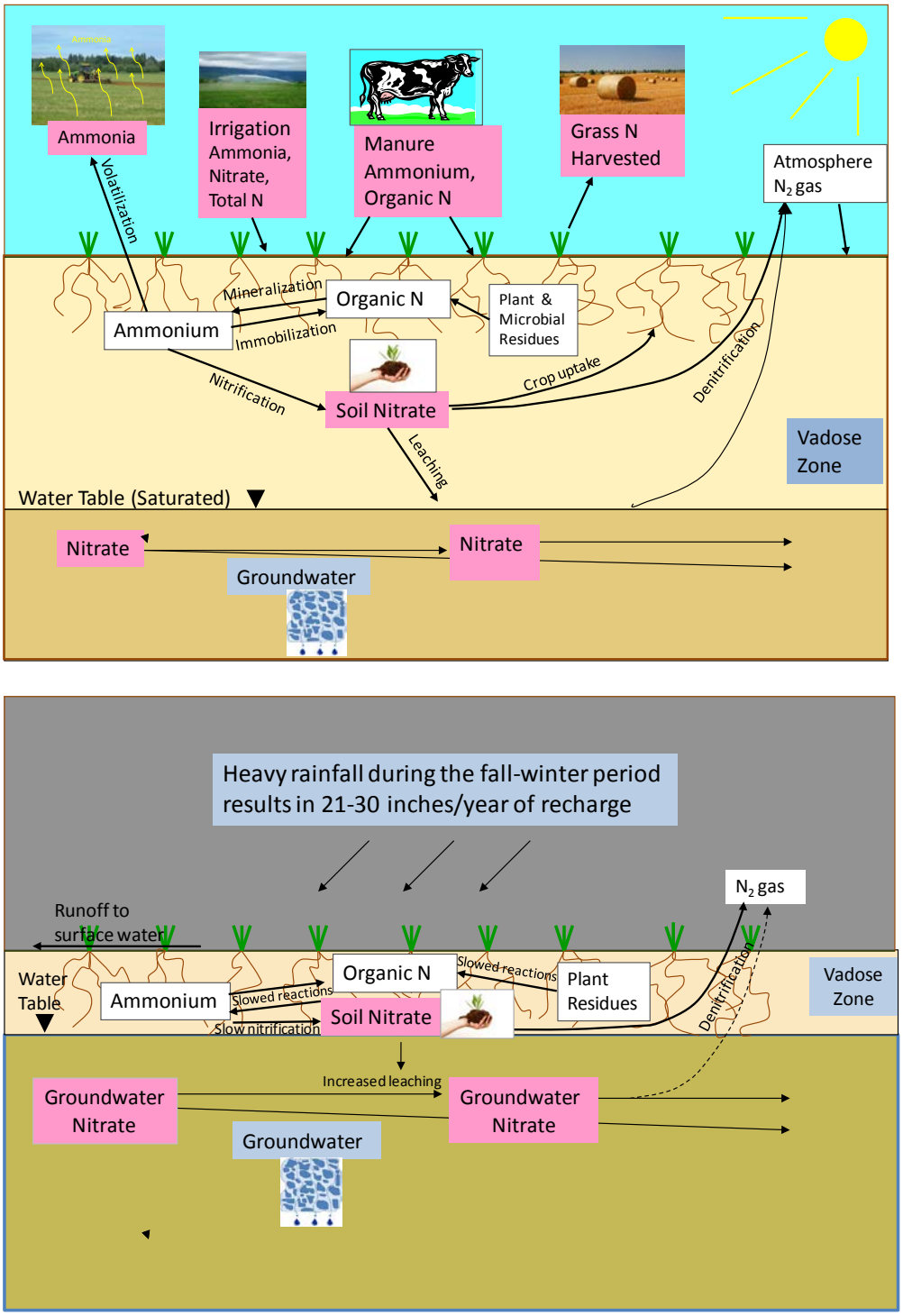

Figure 2. Major nitrogen transformations in spring/summer (top) and fall/winter (bottom) at the study site.

Media in the pink boxes were measured during the study.
Recharge estimate is from Cox and Kahle (1999).

Mineralization

Mineralization is the general term for conversion of organic nitrogen (non-plant available) to inorganic nitrogen (plant-available) by bacteria. The rate of mineralization is affected by temperature, soil moisture, and the redox condition of the soil. Roughly one-third to one-half of the organic nitrogen in land-applied manure mineralizes quickly to ammonium, while the more resistant portion converts gradually over time in a decay process (Beegle et al., 2008). This varies widely among manure types.

Mineralization is slower during the winter than during the warmer seasons (Trindade et al., 2001; Zhao et al., 2010; Cookson et al., 2002). The reverse process of mineralization, called *immobilization*, is the result of the uptake of ammonium or nitrate by microorganisms, temporarily making the nitrogen unavailable to crops. When the microbes die, immobilized nitrogen once again becomes plant available through mineralization.

Nitrification

Nitrification is the bacterial conversion of ammonium to nitrate. This is usually a rapid process favored by warm temperatures, adequate moisture, and aerobic conditions. Nitrification occurs throughout the year. The optimum temperature for nitrification in cultured bacteria from soil is in the range of 25 to 30º C; however, studies have shown that nitrification also occurs at colder temperatures typical of winter conditions (Norton, 2008).

Nitrification is limited in very wet and very dry conditions. In the summer, if the soil moisture becomes too low, bacteria become dehydrated and nitrification is severely slowed (Norton, 2008). Saturated winter conditions can also inhibit nitrification due to reduced oxygen.

Although optimal pH for mineralization is considered to be neutral to slightly alkaline, mineralization has been observed in soils with pH as low as 3.0 (Norton, 2008). Recent evidence suggests that mineralization occurs readily in acidic soils where blueberries are grown (Zebarth, 2013). The nitrification rate for Hale silt loam, the predominant soil type at the study site (pH 5.1-6.5), should not be limited by pH.

Crop uptake

Nitrogen is the main limiting nutrient for most crops. Nitrate is the most available form of nitrogen for plant root uptake (Olson and Kurtz, 1982). Positively charged ammonium ions react with negatively charged soil particles (particularly clay particles), keeping ammonium relatively stationary in soil. However, plant roots readily take up ammonium, if available, especially in the spring before nitrification increases. During the winter months, grass crop uptake rates are significantly slower than during the growing season (Hermanson et al., 2000).

Denitrification

Denitrification is the reduction of nitrate under anaerobic (negligible oxygen) conditions by bacteria to nitrogen gas. In soil, anaerobic conditions often occur under saturated conditions, due in part to the reduced availability of oxygen. Denitrification occurs when the rate of oxygen consumption, usually by bacteria, exceeds the rate of oxygen diffusion in the soil.

Denitrification requires the transfer of electrons from a donor such as organic carbon. Dissolved organic carbon is a component of organic material (including manure) and is the common electron donor for the reaction (Green et al., 2008; Desimone and Howes, 1996).

Organic carbon from manure can build up in the soil over time and enhance the denitrification potential in the soil (Hermanson et al., 2000). Manure itself commonly enhances anaerobic conditions by supplying highly labile forms of carbon that stimulate microbial activity and oxygen depletion (Zebarth, 2013). Nitrate can also be reduced either bacterially or chemically where iron or sulfur are the electron donors (Buss et al., 2005).

Like the nitrification process, the reaction rate for denitrification increases with temperature with an optimum in the range of 25 to 35º C (Buss et al., 2005). Rates of denitrification are known to be highly variable over small distances. *Hot spots* are often reported in soils where denitrification rates are much higher than rates in nearby locations (Coyne, 2008).

Denitrification can occur in both the vadose zone and in groundwater. Denitrification in groundwater is most likely when dissolved oxygen and oxidation-reduction potential are low and organic carbon is available (Singleton et al., 2007; Gillham and Cherry, 1978).

Leaching

In the fall and winter, percolating water due to heavy rain in western Washington transports residual nitrate in the soil and carries it past the root zone, through the vadose zone to the water table. Most, if not all, excess nitrate remaining in the soil after the growing season leaches to groundwater in the fall to early winter (October to January) (Beegle et al., 2008; Downing, 2008; Hermanson et al., 2000; Zebarth et al., 1998; Paul and Zebarth, 1997; Kowalenko, 1989 and 1987). Ammonium is either held in the soil or converted to nitrate and is therefore not normally found in the dissolved phase at the water table.

Leaching also occurs in the late winter/early spring, when precipitation exceeds evapotranspiration and plant uptake of nitrogen is low (Chesnaux et al., 2007; Zebarth and Paul, 1997). Trindade et al. (2001) found high nitrogen mineralization rates during the winter when soil temperatures were above 5°C and soil moisture was near field capacity. If not taken up by plants, this newly generated nitrate can be readily transported to the water table. Several recent studies indicate that winter nitrogen processes are more important than previously thought (Zebarth, 2013).

Leaching during the summer due to irrigation and preferential flow was not addressed in this study but could be a significant factor in the annual nitrogen cycle at the field (Nimmo, 2013).

Dairy nutrient management plans

The Washington State Dairy Waste Nutrient Management Act[5] of 1998 requires that all dairies develop DNMPs. The primary objective of the law was to ensure that surface water and groundwater quality in the state are not adversely affected by dairy manure. DNMPs were required to be approved by July 1, 2002, and implemented with final certification by December 31, 2003. These plans were submitted to local conservation districts for review and approval.

The Washington State Department of Agriculture is responsible for overseeing dairies and DNMPs. The minimum elements of a DNMP are described in the Department of Agriculture web site: http://agr.wa.gov/FoodAnimal/Livestock-Nutrient/DairyNutrientMgmtPlans.aspx

A primary goal of the DNMPs is to balance nutrient application and plant uptake on each individual farm. One aspect of this goal is that the amount of nitrogen removed in the crop match as closely as possible the amount of nitrogen available from the combination of manure nitrogen and nitrogen released from organic material in the soil.

DNMPs are required to outline steps necessary to ensure proper handling and use of dairy manure. Because most of the land on a dairy farm is manured fields, the focus on nutrient management is vital for addressing groundwater nitrate issues. This is particularly true in areas of known vulnerability to groundwater nitrate contamination, such as the Sumas-Blaine Aquifer.

DNMPs require that one composite fall-season soil nitrate analysis be taken at each field receiving dairy nutrients soon after the last harvest and before significant precipitation. Results of the fall (post-harvest) soil nitrate test are used to evaluate the balance between the amount of nitrogen available to the plant (from nutrient application as well as from the soil) and the amount removed by the crop (Sullivan and Cogger, 2003). Sullivan and Cogger established that changes need not be made to current management practices of manured fields if post-harvest soil nitrate is below 15 mg/kg (55 lb/acre typically).

The post-harvest soil nitrate target (15 mg/kg) was based on results of Washington State University studies in and near Puyallup, Washington (Sullivan, 2013) and was not designed to address impacts of nitrate leaching to groundwater. These studies indicated that maximum yields could be attained with 10 mg/kg fall soil nitrate. However, because median post-harvest soil nitrate results for grass fields in Whatcom County were in the 20 to 25 mg/kg range, a compromise of 15 mg/kg was chosen for the target.

The timing of fall soil nitrate sampling can have a critical effect on nutrient balance evaluations. For fine-grained soils in Whatcom County, such as Hale soils at the study site, the recommended time for post-harvest soil nitrate testing is after the last harvest and before 5 inches of precipitation has fallen after September 1. Results typically represent only what is left after at least a portion of the residual nitrate has leached below the sample depth (one foot). If sampling occurs before the last manure application for the year, the result would not include a potentially significant amount of nitrate that will be available for leaching during the rainy season.

[5] Chapter 90.64 RCW

Hirsch (2007) found that soil nitrate sampling after harvest did not capture all of the nitrate leached below the root zone. She recommended testing soil nitrate at the same time as harvest to avoid missing leaching losses. The potential for wide variability in soil nitrate results over short time spans suggests that the standard practice of collecting a single fall soil nitrate sample is a potentially poor predictor of the amount of nitrate that will ultimately reach the underlying water table.

Study Purpose and Objectives

The Bellingham Field Office of Ecology's Water Quality Program requested that Ecology's Environmental Assessment Program design and conduct a long-term study at a dairy farm to:

- Improve our understanding of nitrogen dynamics and fate at a manured field.
- Evaluate the effectiveness of the land application guidelines of Dairy Nutrient Management Plans (DNMPs) in protecting the quality of the Sumas-Blaine Aquifer (SBA) at one location.

The long-term study was designed to measure and document the sequence of changes in nitrogen at a grass field receiving manure in terms of soil, grass crop, manure, and groundwater over 4 years. The dairy operation associated with the study field had an approved DNMP. Monitoring nitrogen dynamics over multiple years allowed variations in weather, manure application, and crop stage to be taken into account.

The primary technical objectives of the study were to:

- Closely track nitrogen dynamics and fate at a grass field overlying the SBA that receives routine manure applications:
  - Quantify the sources of nitrogen input and output to the study field on an annual basis.
  - Quantify changes in soil nitrate and groundwater quality conditions in the study field over time, with a particular emphasis during the period at the end of the growing season.
  - Compare soil and groundwater results to state guidelines and standards.
- Identify the key farm management and environmental factors that influence groundwater and soil nitrate conditions.
- Perform an annual nitrogen mass balance evaluation for the study field.
- Use the mass balance and soil nitrate sampling results to estimate the annual nitrogen residual mass available for leaching to groundwater.
- Evaluate the correlation of the nitrogen residual estimates to the groundwater results, and determine the reliability of these estimates for predicting groundwater quality responses to manure management practices.
- Evaluate if current guidelines for manure management and soil monitoring are adequately protective of groundwater.
- Evaluate the study findings in the context of the SBA as a whole.

Location Description

Setting

The study site is a 22-acre grass field located in northwestern Whatcom County, Washington about 3 miles north of the town of Lynden and 0.3 mile south of the Canada border (Figure 1). The site lies on the flat Lynden Terrace, a glacial outwash plain that slopes gradually southward to the Nooksack River. Bertrand Creek, a perennial tributary of the Nooksack River, lies about 200 feet west of the western boundary of the site. The site is drained by surface ditches and waterways. The site elevation is approximately 130 feet (NAVD88).

Dairy wastewater/nutrients (hereafter referred to as manure) are typically applied as fertilizer on grass and corn fields, which are in turn harvested for livestock feed. Approximately 11 to 14 million pounds of manure nitrogen were applied to fields in Whatcom County in 2010 (Prest, 2011). Larger amounts of manure were applied across the Sumas-Blaine Aquifer (SBA) over the past 40 years, when more dairy cows were present. However, it is not clear how the loading rate (lb/acre) over the aquifer 40 years ago compares with the current loading rate.

Berry growing is also widespread in Whatcom County, in particular raspberry production. Whatcom County produces 65% of the U.S. supply of red raspberries (Whatcom Farm Friends http://www.wcfarmfriends.com/go/doc/1579/181808/). Inorganic fertilizer, the main nitrogen source for berries, is easily leached if not taken up by the crop. Kutchta (2012) found that common irrigation practices for raspberries can result in a large portion of the inorganic fertilizer leaching below the root zone and into groundwater. Other crops grown in the area include blueberries, strawberries, seed potatoes, and nursery stock.

On the Canadian side of the aquifer, poultry production and berry crops are intensive agricultural activities. Poultry production includes land application of manure. Inorganic nitrogen is also applied to berries in British Columbia.

Manured dairy fields lie to the east and west of the study field. However, directly upgradient (north) of the field lies a residence on a 3.5-acre lot. Groundwater beneath another property just east of the upgradient residence may also flow towards the study site during portions of the year, depending on the groundwater flow direction. Both residences were constructed in the past 10 years and are served by on-site sewage systems.

Climate

The regional climate is humid maritime with mild temperatures and rather high precipitation due to proximity to the Pacific Ocean. The Cascade and Rocky Mountains east of the site protect the area from cold air that otherwise would blow down from Canada. The mountains cause moisture rising off the ocean to drop 32 inches/year of precipitation in the southwestern part of the SBA. Precipitation rates increase to 60 inches/year closer to the mountains in Abbotsford, British Columbia.

The closest official weather station is the Environment Canada station at the Abbotsford, B.C. Airport, 7 miles east and slightly north of the study site. Precipitation is known to increase from southwest to northeast across the region; therefore, we assumed that precipitation at Abbotsford would be higher than that at the study site. Annual precipitation for the 30-year period prior to the study, 1973 to 2003, at Abbotsford Airport was approximately 61 inches (http://climate.weather.gc.ca/prods_servs/cdn_climate_summary_e.html).

Roughly 70% of the annual precipitation in the area typically occurs from October through March, outside most of the typical growing season for crops (Kuipers et al., 2012; Cox and Kahle, 1999). Little rainfall occurs in the summer. Potential evaporation at the Abbotsford Airport is typically over twice the precipitation rate from June through August, indicating a seasonal deficit in the water balance (www.Farmwest.com). Where available, irrigation water is applied to crops in the summer.

Soils

Hale silt loam soil overlies the study site. Hale soils are part of the Lynden-Hale-Tromp grouping that overlies much of the SBA. The subsoil at the site (11-27 inches) is mottled, indicating periodic reducing conditions.

When not artificially drained, the rooting depth for crops in Hale soils is limited by a seasonal high water table of 1 to 2 feet (U.S. Soil Conservation Service, 1992). Other characteristics of Hale silt loam include (U.S. Soil Conservation Service, 1992):

- 5-foot depth.
- Moderate permeability in the top 16 inches (0.6 to 2.0 inches/hour) and very rapid below that (greater than 20 inches/hour).
- Clay content, 10-18%.
- Organic matter content, 3-9%.
- pH, 5.1 to 6.5.

Hydrogeology

Regional hydrogeology

The study site lies in the Fraser-Whatcom Lowlands, also referred to as the Lynden Terrace, a glacial outwash plain that slopes gently south toward the Nooksack River. Repeated glacial advances and retreats during Pleistocene times deposited 1,000-2,000 feet of sediments over the area (Figure 3). Outwash from the last glacial episode, the Sumas Stade of the Fraser Glaciation, left gravel and cobble deposits near the Canadian border. These deposits grade finer southward to sand and some clay layers in the Lynden area (Easterbrook, 1971).

During the past 10,000 years, the Nooksack and Sumas Rivers have eroded and reworked the glacial deposits, resulting in the current flat, terraced flood plain morphology. The river has redistributed both the glacial and alluvial material, leaving gravel deposits in upstream areas as well as sand and silt downstream.

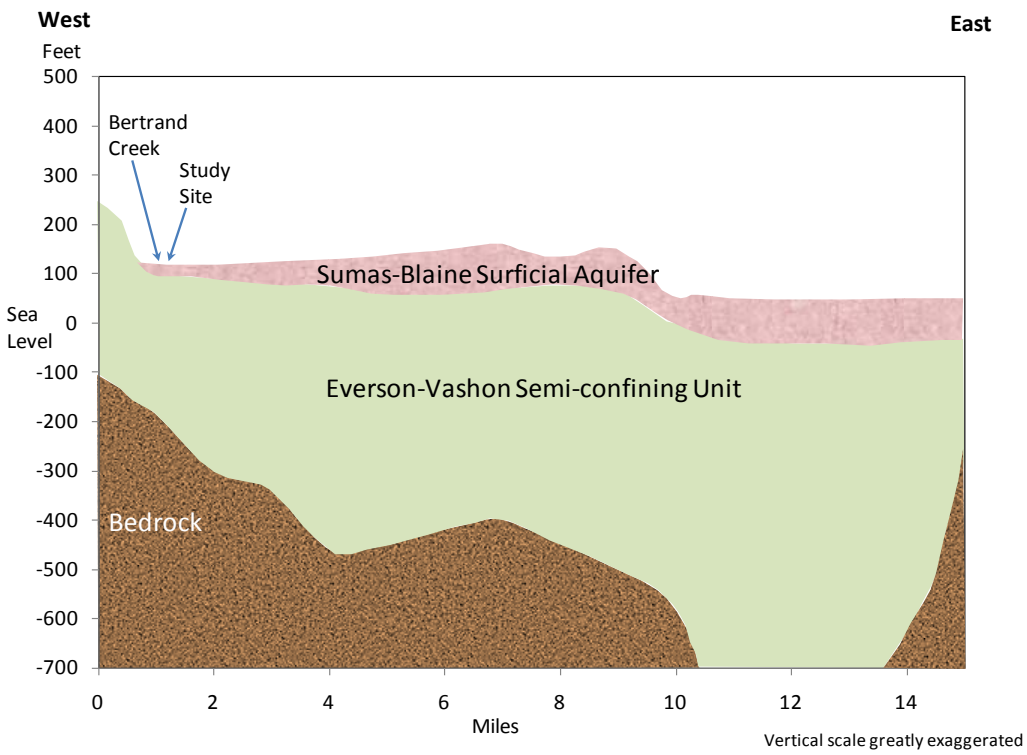

Figure 3. Generalized east-west cross-section near the study area.
Adapted from Cox and Kahle (1999) and Tooley and Erickson (1996).

The principal hydrogeologic units in the study area are shown in Figure 3 and include:

1. Sumas-Blaine Surficial Aquifer
2. Everson-Vashon Semiconfining Unit
3. Bedrock

Sumas-Blaine Aquifer (SBA)

The SBA is about 150 square miles in area and makes up the southern portion of the combined international aquifer system referred to as the Abbotsford-Sumas Aquifer (Figure 1). The SBA consists of stratified, unconsolidated sand and gravel outwash with minor clay lenses. The outwash grades from pebble-cobble alluvium just north of the Canada border in Abbotsford to sand with interbedded fine-grained lenses southwest of Lynden (Cox and Kahle, 1999).

The depth to water is less than 10 feet except for a small portion of the aquifer in the east, making it highly susceptible to surface contamination (Tooley and Erickson, 1996). A system of ditches and tile drains control high water table conditions and facilitate agricultural use in much of the area. Re-routing of a large portion of infiltrating water via tile drains prevents attenuation of leaching nitrate by denitrification and can quickly direct nitrate-rich leachate to surface water (Keller et al., 2008).

The regional groundwater flow direction is generally north to south in the northern part of the SBA (including the study site), toward the Nooksack River (Figure 1). However, local groundwater flow direction can vary (Tooley and Erickson, 1996; Graham, 2013).

The average saturated thickness of the SBA ranges from 25 feet near Blaine in the west to 75 feet near Sumas in the east, thinning at the margins of the alluvial plain (Figure 3). The study site is situated on SBA sediments at the northwestern margin of the plain (Figure 4, Plate 1).

Everson-Vashon semiconfining unit

The Everson-Vashon semiconfining unit is composed of glaciomarine drift consisting of unsorted pebbly clay and sandy silt (Cox and Kahle, 1999). This unit typically functions as a confining bed below the SBA but also includes local coarse-grained, water-bearing lenses as thick as 30 feet. The Everson-Vashon unit is typically 100 to 200 feet thick in the study area and 400 to 700 feet thick in the central axis of the aquifer (Figure 3). High groundwater ion concentrations and difficulty locating coarse-grained lenses preclude the Everson-Vashon unit from consideration as a reliable water supply. The confining layer also prevents significant transport of nitrogen to deeper zones.

Bedrock unit

The bedrock unit underlying the Everson-Vashon semiconfining unit consists of sandstone, mudstone, conglomerate, and coal of the Huntingdon and Chuckanut Formations (Figure 3) (Cox and Kahle, 1999; Creahan and Kelsey, 1988). This unit is not widely used for water supply due to depth and variable water-bearing properties. However, Cox and Kahle (1999) found records for 24 water wells that apparently connect with fractures where the unit is closer to the surface.

SBA properties

Hydraulic conductivity of the aquifer sediments varies widely over the SBA. Based on specific capacity estimates from driller's logs, Cox and Kahle (1999) reported horizontal hydraulic conductivity values in the SBA ranging between 7 and 7,800 feet/day, with a median value of 270 feet/day. Although hydraulic conductivity values varied dramatically over short distances, higher values tended to occur near the Canada border in the northeast part of the SBA. Lower values were measured in the western and southwestern parts of the aquifer. Site-specific measurements of hydraulic conductivity from study monitoring wells are discussed later in this report.

Cox and Kahle (1999) estimated horizontal groundwater velocity throughout the SBA at 0.2 to 29 feet/day based on specific capacity-derived hydraulic conductivity data for 218 wells. For most of the aquifer, they indicated that 2.5 feet/day is a reasonable estimate. Erickson (1991) estimated a groundwater velocity of 1-2 feet/day at a site 2 miles east of the study site using chloride as a tracer. Other velocity estimates for the SBA include 0.3 foot/day 1.8 miles southeast of the study site, based on short-term pumping test results at monitoring wells (Carey, 2002), and 25 feet/day in the coarser-grained Judson Lake area 7 miles east of the study site based on modeling results (Stasney, 2000).

SBA recharge

Recharge of water to the SBA is mainly from precipitation and occurs mostly from September through March, when precipitation exceeds evapotranspiration (Kuipers et al., 2012; Cox and Kahle, 1999).

Maps of recharge estimates by Cox and Kahle (1999) and Kohut (1987) show annual recharge estimates of 16 to 30 inches for most of the SBA, with increasing rates toward the north and east associated with higher precipitation. Recharge in the region is typically 60 to 80% of precipitation (Cox and Kahle, 1999; Malekani, 2012).

Large areas of the SBA are artificially drained to lower the water table below the root zone of crops, which prevents a portion of the infiltrating water from reaching the water table (Cox and Kahle, 1999). Drains typically operate during the winter and early spring, the time when most recharge occurs. The effect of the drain system on regional aquifer recharge rates has not been quantified. Tile drains are not present at the study site.

Study site hydrogeology

The study site is located on the western edge of the SBA, an area dominated by finer-grained material at the surface compared to most of the aquifer. The depth to the bottom of the aquifer is 40 feet at the site, based on well borings from the site (Appendix B, Well AKG726).

Figure 5 (Plate 1) shows the well locations used to develop hydrogeologic cross-sections for the site vicinity (Figure 6, Plate 1). Water level measurements shown on the cross-sections are from domestic and monitoring wells measured during different years; therefore, the water table position is an approximation. Where possible, low water table measurements for the fall were illustrated.

Surface water and groundwater from the site generally flow toward the Nooksack River, 5.5 miles south of the site. Also, localized seasonal reversal of the direction of the horizontal hydraulic gradient in the near vicinity of Bertrand Creek probably occurs, but was not observed in site monitoring wells.

Dairy field management

Regional practices and guidelines

Over the past 20 years, the method of applying manure to crop fields has changed at many farms from mainly large capacity “big gun” spraying and spreading with tanker trucks to methods that apply the manure closer to the soil surface for rapid infiltration or inject into the soil. These newer methods result in reduced loss of ammonia to volatilization and reduced odor, but potentially greater loss of nitrate to groundwater.

Dairies in western Washington typically begin applying manure to forage crops in the spring when weather and soil conditions are conducive to machinery traffic, crop uptake of nutrients is

more active, and the risks of surface and groundwater contamination from bacteria and nitrate are reduced.

Non-application periods for grass fields in non-flood areas of Whatcom County are typically November 1 through February 15 or during periods when the T-Sum value is less than 200 (T-Sum 200).[6] In areas with potential flooding, the non-application period begins October 15 or 30 days prior to the typical flood season (November 15 for the Nooksack River).

The above ordinance also states that, "Should favorable climatic conditions exist, application may begin earlier in the spring than the dates established in this chapter, following approval from the Whatcom Conservation District Board based on T-Sum 200 or best available science. Soil conditions must also be considered when deciding when to apply nitrogen."

The Natural Resources Conservation Service (NRCS, 2005) provides guidance for nutrient management planners regarding conditions for winter manure application. The Whatcom Conservation District is testing a method for incorporating site-specific weather and field conditions into manure application timing and amounts (Application Risk Management) (Embertson, 2010).

Dairy producers may supplement manure applications with commercial inorganic nitrogen fertilizer. Irrigation water is applied on many fields during the dry summer months. Grass crops are typically harvested 4 to 5 times per year.

The treatment system used at the dairy (study site) did not include manure handling techniques to remove solids prior to lagoon storage, which influences the solids content of the manure.

Field management during the study

The 22-acre study field has received manure for over 20 years at about the same application rate as during the study (2004-2009), according to the dairy producer (~400 to 700 lb total N/acre/year). The site was planted in grass before the study and was tilled and re-seeded back to grass in April 2004 using conventional tillage practice. Conventional tillage practices include subsoiling, rototilling, plowing, disking, seedbed preparation, culti-mulching, and planting (VanWieringen and Harrison, 2009). The grass planted in 2004 consisted of 30 lbs Himark Fescue, 10 lbs Quartet Perennial Ryegrass, and 50 lbs Oats.

During the study, the dairy producer managed the field as before the study. The first liquid manure application for each year during the study occurred in February, March, or April, depending on weather and soil conditions. Manure was applied most often using an aerator (also referred to as subsurface deposition) with equipment from Aerway® Aerators & Parts (Figure 7). Tines were set 7.5 inches (19 cm) apart on a roller and allowed to drop 4 inches (10 cm) below the soil surface, creating intermittent slices 5 inches (12.5 cm) deep at the surface. Because tines

[6] The T-Sum value is derived by summing the daily mean temperature (in degrees Celsius) starting January 1 of each year. (Ord. 98-074; Ord. 98-056--Whatcom County Code 16.28.030 www.codepublishing.com/wa/whatcomcounty/frameless/index.pl?path=../html/Whatco16/Whatco1628.html)

wear down over time and are not always fully inserted, slices may be less than 5 inches deep (Clark, 2013). Liquid manure was sprayed over the slices.

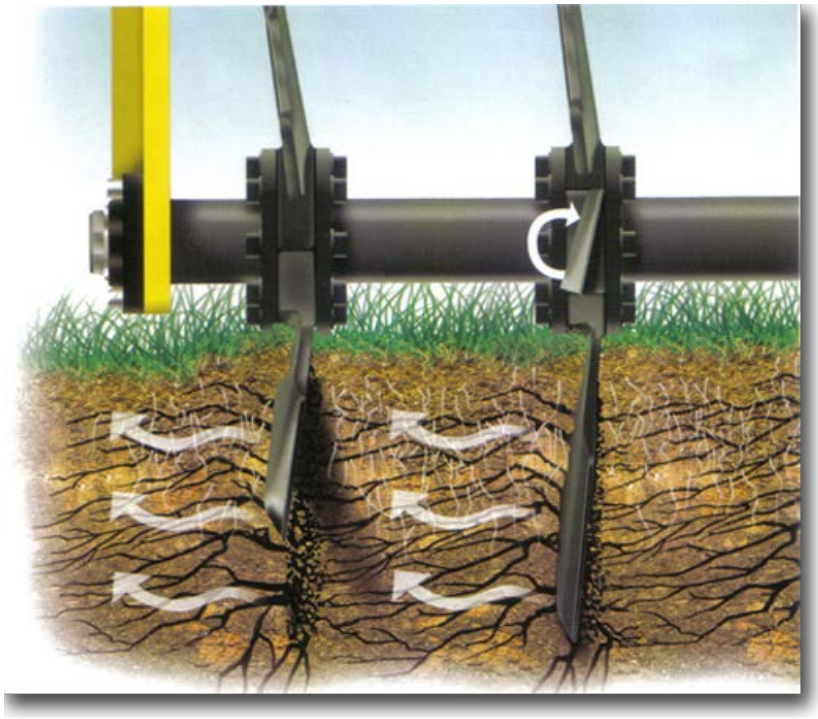

Figure 7. Sub-surface deposition of manure at the study site (top) and close-up diagram (bottom) from Aerway® Aerators & Parts.

Supplemental liquid manure from another dairy was applied by injection on 3 occasions in 2005 and 2008 (VanWieringen and Harrison, 2009). This method injects manure a few inches below the top of the soil.

Manure was typically applied 3 to 5 times per year following each grass cutting. The final manure application for any given study year occurred between the end of August and early October. In 2 of the 4 study years, the final manure application occurred after the last crop harvest (2005 and 2006).

Irrigation water from a nearby shallow well was applied at the study site using a hard-hose reel with a "big gun" sprinkler and pump each summer. The grass crop was harvested 4 to 5 times each year.

Study Design

Ecology's Environmental Assessment Program partnered with Washington State University's Livestock Nutrient Management Program (WSU) to accomplish the project objectives. We conducted the following tasks as part of a multi-media sampling program at the study site:

- Conducted a 4-½-year intensive multi-media monitoring program at a 22-acre manured dairy field.
- Analyzed the study field nitrogen mass balance each year and comparing the estimated nitrogen residual to shallow, underlying groundwater nitrate concentrations.
- Evaluated the effect of various environmental and management factors on the nitrogen mass balance and groundwater nitrate concentrations.

Our monitoring program focused on the following components of the nitrogen cycle to evaluate the balance of nitrogen at the study field system (crop, soil, and shallow groundwater) as shown in Figure 8:

- Inputs
  - Manure and inorganic fertilizer (mass of nitrogen applied to the field)
  - Irrigation water (volume and nitrogen concentration added to the field)
- Outputs
  - Grass harvested (mass of nitrogen removed from the field)
- Residual
  - Soil (fall nitrate concentration and estimated mass in the top 1-foot of soil)
  - Groundwater near the top of the water table (nitrogen concentration)

Nitrogen outputs due to volatilization were included in plant-available nitrogen estimates (Sullivan, 2008) or estimated using literature values. Denitrification in the soil was assumed to be negligible during the growing season (Sullivan, 2008).

To support the evaluation of nitrogen transport at the study site, additional field work was conducted to characterize the hydrogeology and soil characteristics of the study site. This included:

- Measuring static water levels in monitoring wells.
- Conducting tests of the hydraulic characteristics of the unconfined aquifer underlying the study field.
- Conducting grain size analysis of site deposits.
- Measuring chloride in groundwater to use as a conservative (non-reactive) tracer.
- Measuring other water quality constituents in groundwater that contribute to understanding nitrate levels.

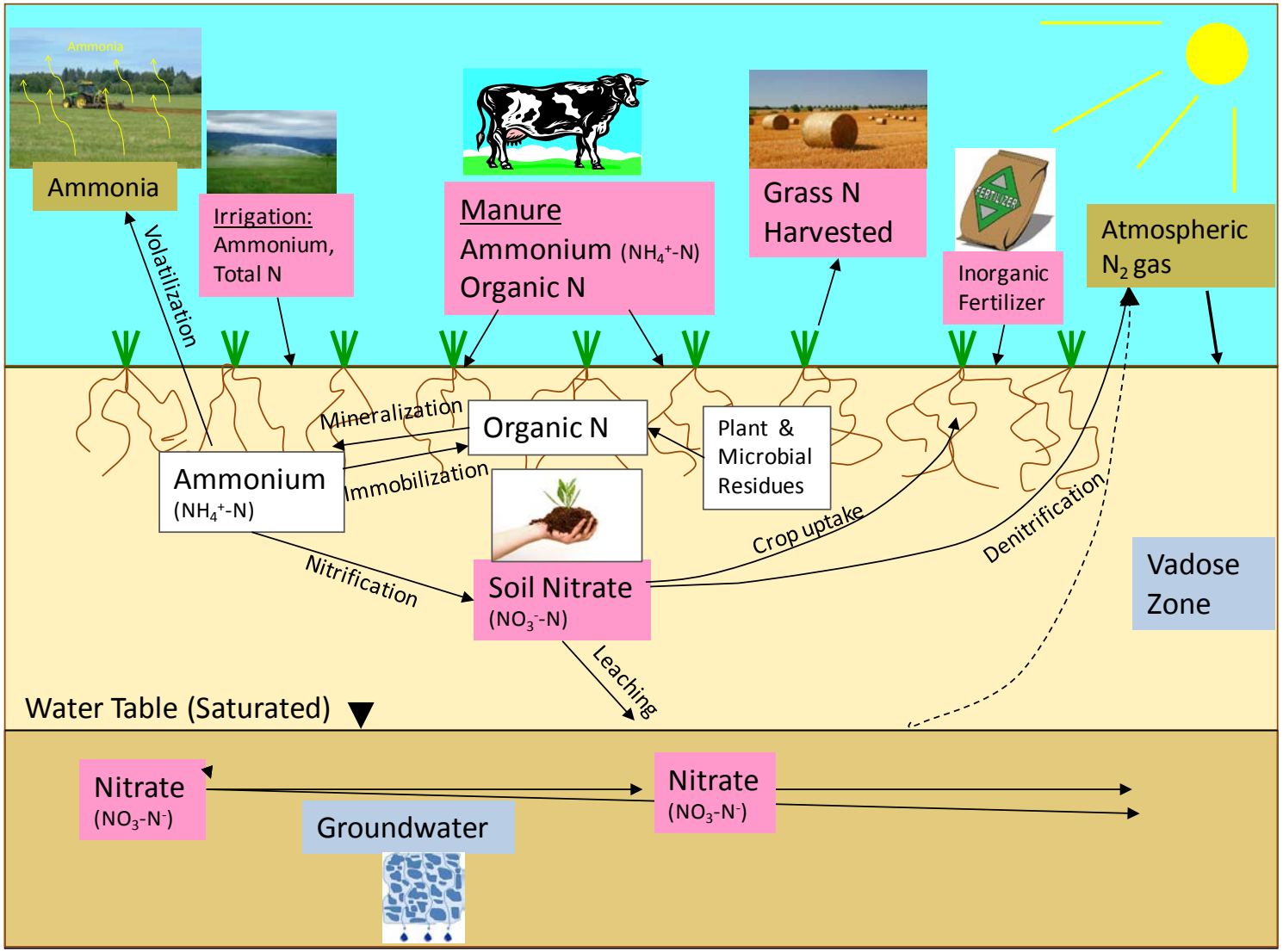

Figure 8. Major nitrogen compartments in the study field.

Media in pink boxes were monitored. Items in brown boxes were estimated.

Further details of the study design are described below and in Carey (2004). Figure 9 shows the analytes sampled for each media. Table 1 lists the analytes sampled and the frequency of sampling.

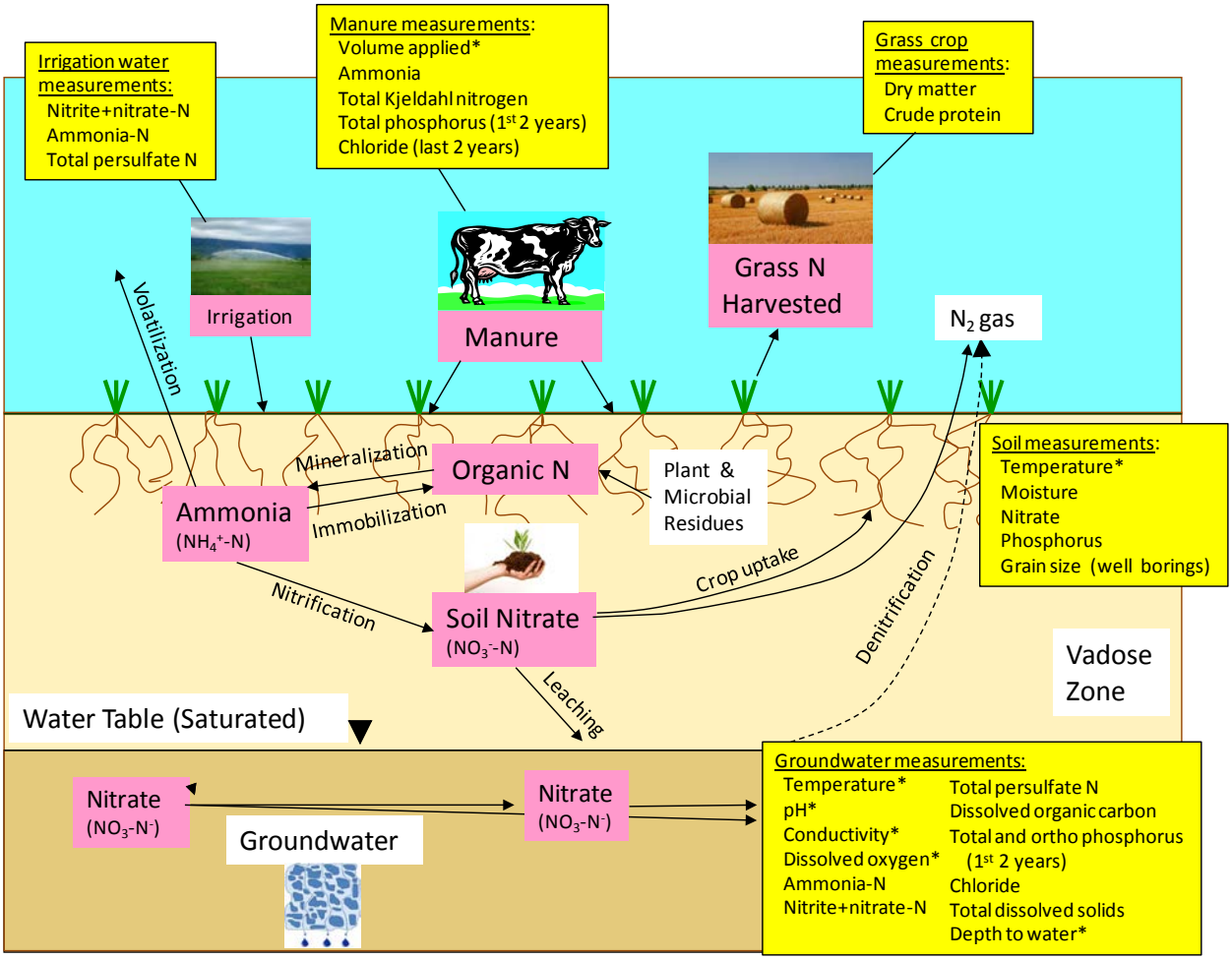

Figure 9. Analytes measured in each media are shown in boxes.

*Analytes were measured in the laboratory, except for those analytes with *s, which were measured in the field.*

Table 1. Analytes measured in groundwater, soil, manure, grass, and irrigation water. *Groundwater samples for laboratory analysis were filtered in the field except for groundwater from the deep well, AKG726, which were not filtered. Analytical methods are listed in Carey (2004).*

Analyte	Matrix[1]	Frequency
Field		
Groundwater temperature	G, S	Monthly (summer every 6 weeks)
pH	G	Monthly (summer every 6 weeks)
Specific conductivity	G	Monthly (summer every 6 weeks)
Dissolved oxygen	G	Monthly (summer every 6 weeks)
Soil temperature	S	Monthly (weekly August-November)
Soil moisture	S	Monthly (weekly August-November)
Laboratory		
Ammonium-N	G, M, I	G—Monthly (summer every 6 weeks) M—Each time manure applied I—Each time irrigation water applied
Nitrite+nitrate-N (Nitrate-N)	G, I	G—Monthly (summer every 6 weeks) I—Each time irrigation water applied
Nitrate-N	S	Weekly August-November, otherwise monthly
Total persulfate N	G, I	G—Monthly (summer every 6 weeks) I—Each time irrigation water applied
Total Kjeldahl N	M	Each time manure applied
Ortho phosphate	G	Monthly (summer every 6 weeks)—(2004-2006)
Total dissolved phosphorus	G, S, M	G—Monthly (summer every 6 weeks) S—Annually M—Each time manure applied
Chloride	G, M	G—Monthly (summer every 6 weeks) M—Each time manure applied
Total dissolved solids	G	Monthly (summer every 6 weeks)
Total organic carbon	G	G—Monthly (summer every 6 weeks)
Grain size	S	One time for drilling samples
Dry matter	Gs	Each time grass crop harvested
Crude protein (N)	Gs	Each time grass crop harvested
Organic matter	S	Annually
Soil chemistry[2]	S	Annually

[1] Matrix codes: G: Groundwater; S: Soil; M: Manure; Gs: Grass; I: Irrigation water.

[2] Soil chemistry: Phosphorus, potassium, boron, zinc, manganese, copper, iron, calcium, magnesium, sodium, sulfur, buffer pH, cation exchange capacity, total bases, base saturation, pH, electrical conductivity, ammonium N.

Methods

Weather conditions

A battery-powered Onset weather station was installed in the field during the study (Ecology weather station). Precipitation and air temperature measurements were recorded every 15 minutes from September 22, 2004 through March 18, 2009.

Nitrogen inputs and related constituents

Manure

Samples of liquid manure applied to the field were collected from the applicator when manure was being applied on 17 out of 18 times that it was applied. Manure was applied 4 to 5 times per year. The standard operating procedure (SOP) for manure sampling is described in Appendix C. The dairy producer reported the amount and timing of inorganic fertilizer application on 2 occasions, one time each in 2006 and 2007.

Irrigation water

Water for irrigation was applied each summer, with a total of 9 events during the study (Appendix D). Samples for volume and water quality were collected each time from the irrigation water applicator while the field was being irrigated, with the exception of 2 events in 2005. The amount of water applied during these 2 irrigation events was estimated by the producer.

Irrigation water samples were collected into 3 acid-washed buckets twice at different times throughout the irrigation event. The volume of water collected in each bucket was measured, and the rate of application was estimated using Equation 1:

$$I = \frac{\left[\frac{V}{T}\right]}{A} \qquad (Eq.\ 1)$$

where:
I = Irrigation rate (inches/day)
V = Volume of water in buckets (cubic inches)
T = Time (day)
A = Area of the buckets (square inches)

Contents of the 3 buckets were then composited and mixed in an acid-washed container. The sample was poured into 2 bottles with preservative and placed in a cooler with ice for shipping via FedEx to Ecology's Manchester Environmental Laboratory.

In addition to nitrite+nitrate -N, which was analyzed throughout the study, irrigation water samples collected on September 12, 2007 and for both applications in 2008 were also analyzed

for ammonia-N and total persulfate N. "Nitrate+nitrite-N" is referred to as "nitrate-N" in this report, because nitrite-N is typically negligible in surface water and groundwater (Sawyer and McCarty, 1978).

Nitrogen outputs – grass crop

Grass samples were collected from 2-ft by 2-ft squares for yield estimate and quality analysis 1 to 2 days before each crop harvest, which occurred 4 to 5 times each year. Five subsamples were composited to form one sample on each date. The process was repeated for a duplicate sample.

The 10 general subsample locations (5 for the sample and 5 for the duplicate) were initially randomly selected and recorded using a Global Positioning System (GPS). The same 10 locations were then sampled each time thereafter. The SOP for grass sampling is described in Appendix E.

Soil conditions

Temperature

Soil temperature was also measured at a 6-inch depth during each soil sampling event. Soil temperature was measured by inserting 2 temperature probes 6 inches into the ground near the first soil coring location and averaging the 2 results.

Soil nitrate, moisture, organic matter, and other constituents

The frequency and timing of soil sampling events was designed to correspond with the likelihood of excess soil nitrate leaching to the water table. From December through July of each study year, when crops are growing most rapidly, precipitation rates are low, and reduced leaching of nitrate to groundwater is expected, we sampled soils on a monthly basis. From August through November, when the potential for leaching of residual nitrate from the soil column is of greatest concern (as precipitation begins to exceed evapotranspiration, and crop uptake rates declines), we sampled soils weekly.

The SOPs for soil sampling are summarized below and are based on methods described by Sullivan and Cogger (2003). The complete SOPs are described in Appendix F. Soil sampling locations are shown in Figure 10.

Each sample consisted of a composite of 15 soil core subsamples. A one-inch diameter hand-held coring device was used to collect each 1-foot deep soil core subsample at initially random locations around the field. The location of each core was verified using a GPS. The same locations were re-visited and sampled each subsequent event.

Loose crop or manure residue at the top of each core was discarded. The remaining soil from each of the 15 cores was placed in a 5-gallon bucket and mixed thoroughly by hand with a properly decontaminated trowel. The composite sample was then divided into 2 to 3 subsamples and placed in clean plastic bags, one for analysis at the contract laboratory, one for archival storage at WSU-Puyallup. One composite split sample was sent to a contract lab each year.

Figure 10. Soil sampling locations.

All sample sites were in the study field, although the aerial photo does not provide this level of accuracy.

Two sets of soil cores at 15 sites were used to obtain 2 samples of composited soil on each sampling date. A duplicate set of soil cores was collected each sampling day at a different set of 15 locations than were initially randomly selected. The duplicate sample was handled the same as the first sample. Subsequent soil cores were collected within a few feet of the original 30 duplicate locations.

Groundwater conditions

Monitoring well installations

Nitrate enters groundwater beneath the study field via 3 major pathways:

- Leaching and infiltration of nitrate from overlying soils through the vadose zone.
- Direct dissolution of nitrate in soil when the water table rises and saturates the lower portion of the soil column.
- Lateral groundwater transport of nitrate from upgradient of the study field.

Because of the high solubility of nitrate, infiltrating water can rapidly transport dissolved nitrate through the root zone and eventually to the water table. This process is of particular concern during periods of heavy precipitation (fall/winter/early spring), when the water table rises 5 to 9 feet in elevation, minimizing the transport distance to groundwater. This leaves little chance for nitrate to remain in the thin unsaturated soil layer (Kowalenko, 1989 and 1987; Zebarth et al., 1996).

Because the most recently recharged groundwater is closest to the top of the water table (Figure 11; Wassenaar et al., 2006), we completed the monitoring wells in a manner to intersect the water table and characterize recent recharge.

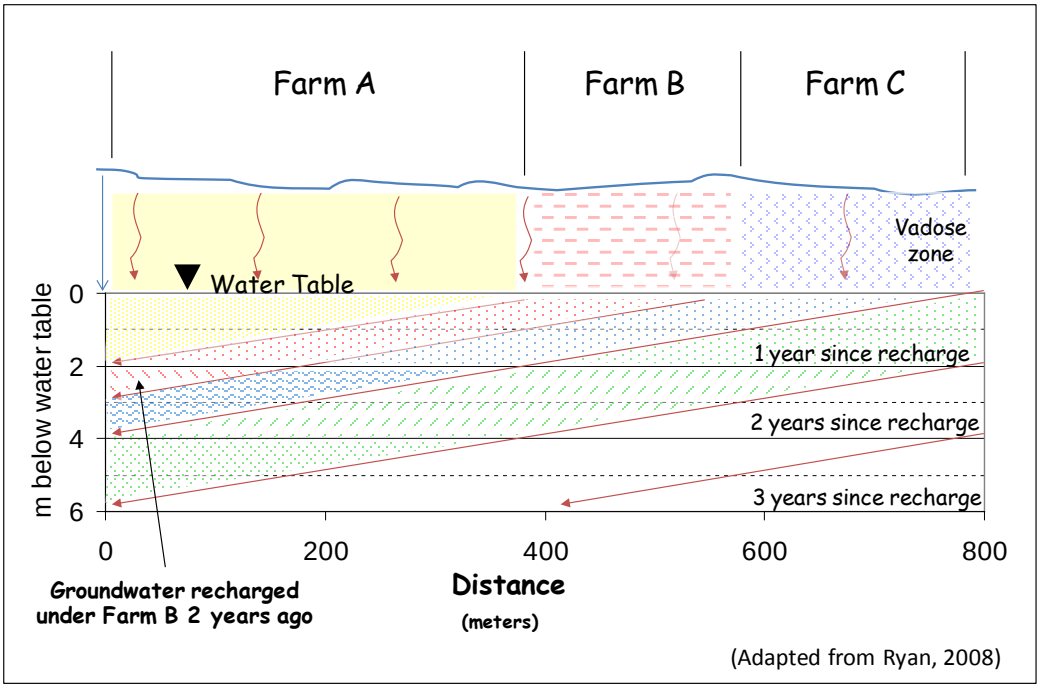

Figure 11. Generalized groundwater flow beneath fields in the Abbotsford area 7 miles northeast of the study site, showing that water near the top of the water table represents the most recent recharge from above.

Adapted from Ryan (2008).

The monitoring well network consisted of 6 shallow wells and 1 deep well in the study field. Two shallow wells were installed in 3 rows from upgradient to downgradient in the field (Figure 12). All monitoring wells were within the manured field (see Appendix B for drilling logs). The monitoring well locations and construction specifications were chosen to:

- Describe the subsurface hydrostratigraphy and hydraulic properties.
- Estimate the groundwater flow rate and direction.
- Obtain samples representative of the most recent groundwater entering the aquifer on the site (top of the water table).

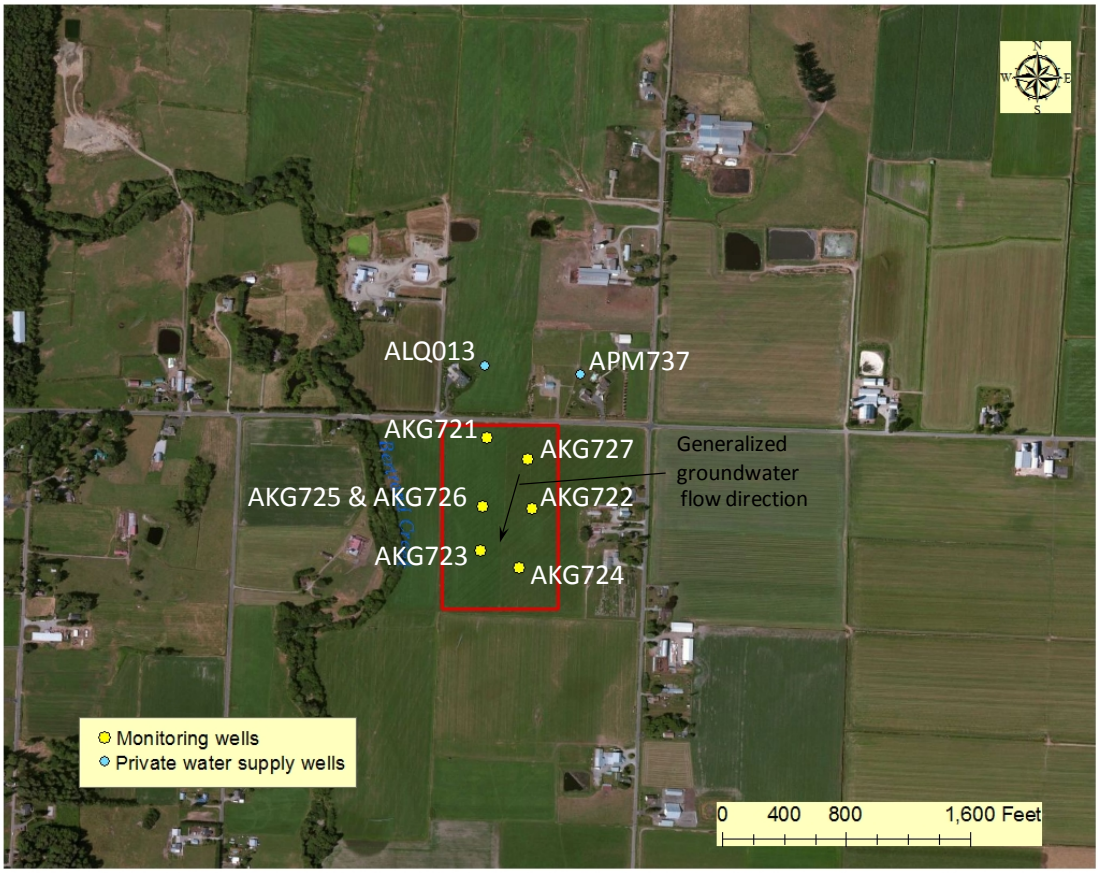

Figure 12. Location of wells sampled during the study.
All wells are in the study field.

Monitoring wells were installed by Holt Drilling, Inc., Puyallup, Washington, using a 4¼-inch inside-diameter hollow stem auger (8-inch outside-diameter). The wells were installed from August 25 to 26, 2004, about 4 weeks before groundwater samples were collected. Six wells were 12 to 13 feet deep, and one well drilled to the bottom of the unconfined aquifer was 38 feet deep. See Appendix G for a summary of well locations and construction information.

The monitoring wells were constructed according to the state standards for resource protection wells.[7] Figure 13 shows the standard construction plan for the shallow monitoring wells. The deep well was constructed similar to the Figure 13 plan except that the well screen was 10 feet long. The depth to the top of the screen was approximately 5 feet below ground surface in the shallow wells and 28 feet in the deep well.

Shallow wells were constructed with 2-inch inner diameter polyvinyl chloride (PVC), flush-threaded casing, and commercially fabricated 7-foot long screens (10-foot for the deep well) with a slot size of 20. We selected 7-foot long screens to provide as close to year-round access as possible to the top of the water table, which fluctuates roughly 7 feet over the year.

[7] Chapter 173-600 WAC

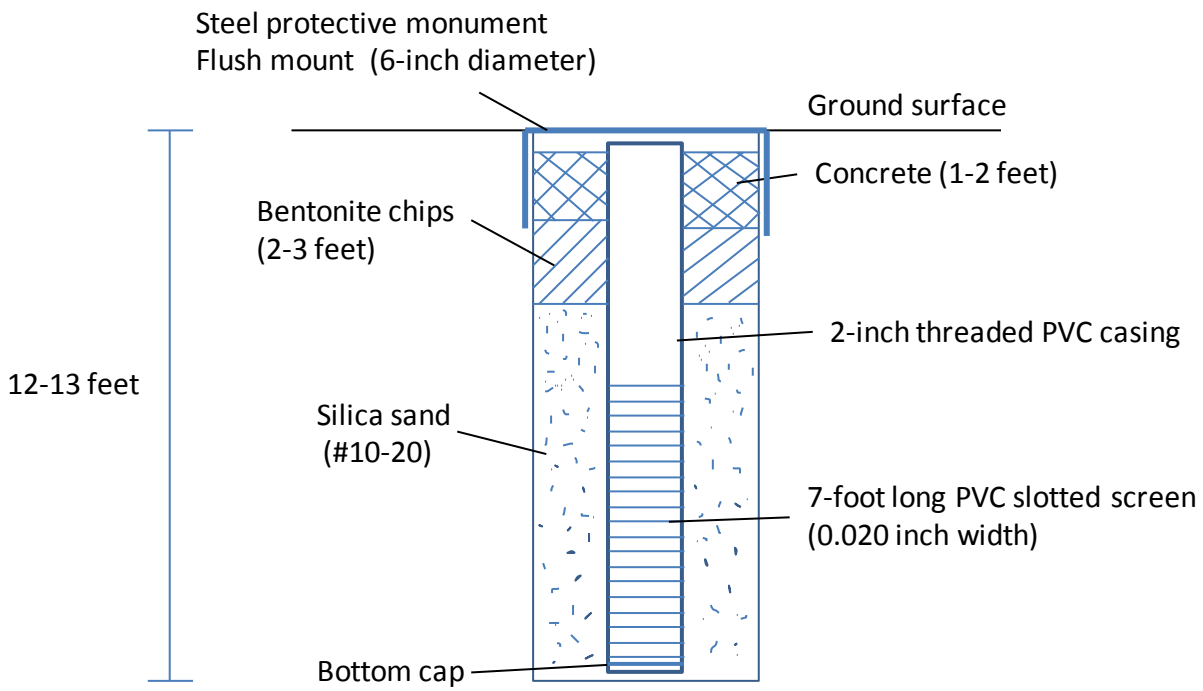

Figure 13. Schematic for shallow monitoring well construction.

The sand pack consisted of 10-20 silica sand installed continuously over the screened interval to 1 to 2 feet above the top of the screen. Bentonite pellets were placed within the annular space between the boring and the PVC casing from the top of the sand pack to within 1 to 2 feet of the surface. Concrete was installed around the top 1 to 2 feet of casing.

Split spoon core samples (18 inches long) were collected at 5-foot intervals during drilling. Core samples were placed in clean, labeled, plastic zip-lock bags. Fifteen split spoon samples were analyzed from the 7 wells for grain size according to ASTM Method D422 (ASTM, 2003). Sample intervals were selected to cover the range of material types encountered and the range of depths. Triplicate samples were analyzed for the deepest sample, 40 feet, in AKG726. The texture of the 40-foot sample changed dramatically from fine sand above 40 feet to clay and silt, indicating the base of the Sumas-Blaine Aquifer.

Each monitoring well was equipped with a water-tight cap and lock. A steel 6-inch diameter flush-mount outer protective casing was installed over the PVC well. The steel casing extended to a depth of 2 feet below ground.

After completion, the wells were developed by the driller using a jetting technique until the water removed from the borehole was free of sediment. A state well tag with a unique ID number was attached to each well.

Upgradient private wells

In addition to groundwater monitoring wells installed in the manured field, we sampled 2 upgradient private water-supply wells in 2008, one north of the study site and one northeast of the site (Figure 12). Both private wells are roughly 380 feet from the boundary of the study site.

Well ALQ013 was sampled 2 times (March 11 and April 2) and well APM737 one time on March 11 (see Appendix B for drilling logs). The wells are screened at 29 to 34 feet depth. Analytes included temperature, pH, specific conductivity, dissolved oxygen, ammonia-N, nitrate-N, total nitrogen, chloride, and total dissolved solids. Samples from these wells were not filtered. See Marti (2011a) for the SOP used to sample upgradient wells.

Hydraulic testing

We conducted aquifer hydraulic testing to determine if the subsurface hydraulic properties at the study site are similar to those reported for the Sumas-Blaine Aquifer as a whole. Hydraulic testing helps estimate sediment permeability and groundwater velocity, which affect how quickly nitrate and other dissolved constituents move once they reach the water table.

Short-term specific capacity tests were conducted on April 4, 2006 at 3 monitoring wells to provide an approximate estimate of the hydraulic properties of the aquifer materials. We tested the deep well (AKG726) and 2 shallow wells (AKG723 and AKG725) to characterize the shallow and deeper portions of the unconfined aquifer.

A specific capacity test consists of pumping a well at a known rate until the water level in the well equilibrates. The drawdown is recorded throughout the test period and is used with the well construction information to estimate the transmissivity and hydraulic conductivity of the aquifer in the vicinity of the well screen.

Specific capacity refers to the rate of well discharge divided by the drawdown in the well and is measured in gallons per minute per foot of drawdown. Bradbury and Rothschild (1985) developed a technique for estimating hydraulic conductivity using specific capacity based on the Theis (1963) graphical method. The Bradbury and Rothschild method uses a computerized iterative procedure to estimate transmissivity, which is then converted to hydraulic conductivity by integrating over the saturated thickness. The method uses the Cooper-Jacob approximation of the Theis equation with corrections for partial penetration and well loss (turbulent flow in the well during the test). See Appendix H for method details.

The assumptions of the Bradbury and Rothschild (1985) technique include:

- Confined, non-leaky, homogeneous, isotropic aquifer
- Storage coefficient is known
- Minimal well loss
- Penetration of the aquifer is known

Despite not meeting the assumptions of confined conditions, Bradbury and Rothschild (1985) had success using the computerized method in unconfined sand and gravel wells of the Central Sand Plain of Wisconsin. They found close agreement between results of full-scale pumping tests and specific capacity tests in individual wells. Sinclair (2002) likewise found close agreement between hydraulic conductivity results from large-scale aquifer tests in the Sequim-Dungeness area of Washington and results from specific capacity tests.

Groundwater sampling

The field and laboratory methods used for groundwater monitoring are described in Carey (2004). Standard protocols used in Ecology's Environmental Assessment Program were followed for measuring field parameters and collecting samples for laboratory analysis. Likewise, standard methods were used for sample handling, preservation, and storage (Marti, 2011b).

Groundwater samples were collected monthly during the fall and winter, and every 6 weeks in the spring and summer. Prior to sampling, water levels were measured at each well using a clean, calibrated electric probe per methods described by Marti (2009). Measurements were recorded to 0.01 foot and are considered accurate to 0.03 foot.

For well purge and sampling, we used a peristaltic pump with dedicated high density polyethylene tubing that remained inside the well between sampling events. A short section of silastic tubing at the pump head was used for all shallow wells and was replaced for each new sampling event. The pumping rate for the shallow wells was approximately 0.11 gallon/minute. The intake for the sample tubing was set at 1.5 feet below the top of the water table, or at the top of the screened interval when the water table was above the screened interval.

We purged and sampled the deep well (AKG726) using a submersible pump with dedicated polyethylene tubing. The pumping rate for purging and sampling at well AKG726 was approximately 1 gallon/minute.

We purged each shallow well for a minimum of 20 minutes and until field parameters (temperature, pH, conductivity, and dissolved oxygen) stabilized to within 10% for consecutive measurements spaced 5 minutes apart. The deep well was purged for 3 to 5 minutes, because the flow rate was high enough that field parameters stabilized quickly. Field parameters were measured inside an enclosed flow-through cell to minimize atmospheric bias effects (Figure 14).

Samples from the shallow wells were field-filtered using dedicated, in-line 0.45 um filters. After discarding the initial 50 milliliters of filtrate, samples were collected in clean bottles obtained from Ecology's Manchester Environmental Laboratory, as shown in Figure 15.

Samples collected from the deep well were not filtered. The higher discharge rate from the submersible pump used for the deep well made it more difficult to use field-filters. The discharge from the deep well was visually clear, and we assumed that the constituents of interest would not be in the particulate form.

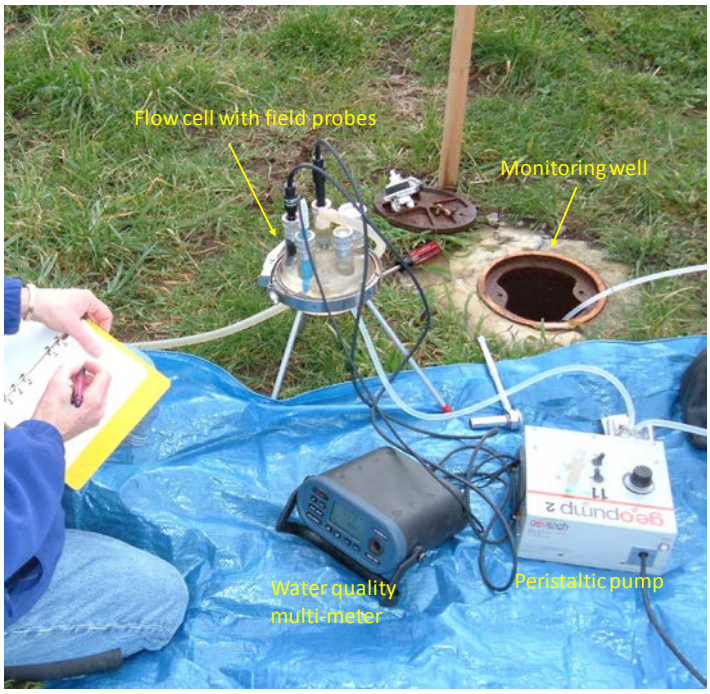

Figure 14. Groundwater sampling flow cell, peristaltic pump, and multi-meter for measuring field parameters: temperature, pH, conductivity, and dissolved oxygen.

Figure 15. Field-filtering a groundwater sample using a disposable, in-line filter that bypassed the flow cell.

Project Quality Assurance

Results of quality assurance testing for each media sampled are described in detail in Appendix I and summarized below. Overall, the results of the quality assurance testing indicated that the analytical data collected during the study are of good quality and can be used without qualification. In a few cases, qualifiers were added to a data result to identify values that may be outside of the project data quality objectives.

Manure

Duplicate manure samples were collected at least once each year and analyzed for percent solids (dry matter content), ammonium, and total nitrogen. See Appendix I, Table I.1, for results.

The range of relative standard deviations (RSD) for ammonium in duplicate manure samples was 0.08 to 30.4% with a mean of 8.0%. The project data quality objective RSD of 7% established in Carey (2004) was met on 4 out of 6 occasions.

The range of RSD for duplicates of manure total nitrogen samples was 2.57 to 18.9% with a mean of 8.0%. The target RSD of 7% was met on 3 out of 5 occasions. Manure results for dates that did not meet the target precision are qualified in the results (Appendix L).

Grass crop

Duplicate grass samples were collected each time the field was harvested from July 17, 2005 through October 21, 2008 (Appendix I, Table I.2). Individual samples were collected for the 3 harvests prior to July 17, 2005.

Eighty-three percent of wet-weight RSD values were within the 10% target (Appendix I, Table I.3). Eighty-nine percent of the dry-weight values and all of the crude protein values were within the 10% target range. Values outside the target range are qualified in Appendix I, Table I.2, and in the results (Appendix N). Based on our experience, the 10% target for RSD of grass parameters is particularly stringent.

Soil

Split soil samples were collected quarterly except in 2007, when split samples were not collected. A split sample consisted of a portion of 1 of the 2 duplicate samples for a given date. The split samples were analyzed by Soiltest Farm Consultants in Moses Lake, Washington.

The RSDs for 14 split soil nitrate samples are shown in Table I.4. The mean RSD was 11%, above the target of 7% (Carey, 2004).

Results for duplicate soil nitrate samples are shown in Table I.5. The mean RSD for 107 duplicates was 13.1%. Thirty-nine percent of the RSDs for duplicate soil nitrate samples met the 7% target precision.

The target precision for soil nitrate set prior to the start of the study may have been unreasonably low. The average RSD for 13 soil nitrate studies at dairy farms conducted by Washington State University was 16% (Bary, 2010). The range in soil nitrate concentration values for most of these studies was less than that in the current study.

Because the target precision for soil nitrate was so much lower than the average RSD in similar studies, and the range of soil nitrate values in the our study was so wide, a more realistic threshold for acceptable precision is 20%.

Soil nitrate duplicate samples with RSDs less than 20% are considered acceptable for use without qualification (Bary, 2012). Twenty-four of 107 soil nitrate duplicate samples exceeded the 20% threshold for RSD and are qualified in Table I.5 and in the results (Appendix K).

Groundwater

Field quality assurance

All groundwater field meters were calibrated at the start of each day according to the manufacturer's instructions. Replicate field measurements were collected at one monitoring well for each sampling event (replicating a different well each round) to assess overall precision of field and laboratory results (including the environmental variability over a few minutes between samples). Replicate samples were collected by immediately repeating the normal sampling process at the chosen well. Replicate samples were submitted blind to the laboratory.

The relative percent difference (RPD) for field parameters excluding dissolved oxygen was 0.6-1.4% (Appendix I, Table I.6.). The RSD for dissolved oxygen, which was often in the 0-3 mg/L range, was 8.7% (Table I.7).

The RPD values for water quality parameters represent combined field and laboratory precision. The target precision for nutrients, 7% RSD, and for chloride and dissolved organic carbon, 10% RSD, were met in most cases. Values that did not meet the targets are qualified in Appendix I., Table I.6.

Mean RPDs for laboratory analytes based on field replicates ranged between 2.2 and 5.3%, excluding total dissolved phosphorus (including nitrate-N, total persulfate nitrogen, ortho-phosphorus, chloride, dissolved organic carbon, and total dissolved solids). See Table I.7. The mean RPD for total phosphorus was 17%. Total dissolved phosphorus is the only analyte that did not meet the target precision.

On 6 occasions in 2008, a blank sample of de-ionized water from Manchester Laboratory was collected using the same silastic tubing for the peristaltic pump that had been used for sampling the monitoring wells. Results of blank samples were used to evaluate potential cross-contamination between sample locations from the silastic tubing. Most of the blank results for the nitrogen series were below detection (Table I.8). On May 6 and June 19, 2008, both nitrate-N and total persulfate nitrogen (TPN) were detected at concentrations roughly 1% of sample values. These results indicate that using the same silastic tubing when purging and sampling

each well (one new piece of silastic tubing for each sampling event) was not a significant cross-contamination source.

Laboratory quality assurance

Laboratory quality assurance consisted of duplicate blanks, duplicate samples, spiked samples, and check (control) standards. Manchester Laboratory conducted internal quality assurance reviews. Most of the results are considered acceptable for use without qualification. Some data were qualified as described in Tables I.6 through I.8.

Impact of manure leakage on groundwater quality results

On February 7, 2005, the sealing well cap and flush-mount monument cover for monitoring well AKG727 were inadvertently left off the well. The field received one manure application before the error was discovered on March 3, 2005. The well was purged, resealed and remained closed for the rest of the study except when sampling. A 2-dimensional analytical model was used to estimate the potential impacts manure entry into the well had on the groundwater quality during the study (Appendix I.9).

The model assumed that manure was constantly injected into well AKG727 for 2 days. The resulting groundwater concentrations in well AKG727 and all downgradient wells were estimated using the model. Any estimated groundwater nitrate-N concentrations affected by more than 0.25 mg/L-N were disqualified. Only data from AKG727 were affected by 0.25 mg/L-N and greater. Potentially affected water quality data collected from February 7 through July 14, 2005 at AKG727 were not used in any data analyses in this report.

Results

Precipitation

The on-site weather station recorded precipitation and air temperature measurements on 75% of the days from September 22, 2004 through March 18, 2009 (1,262 days). Battery problems prevented data collection on 25% of days (418 days). Daily precipitation data at the study site for 1,072 days correlated with daily precipitation data from the Abbotsford, British Columbia Airport ($r^2 = 0.764$)[8]. The relationship between precipitation at the 2 sites was estimated using Equation 2:

$$y = 0.707\,x + 0.003 \qquad (Eq.\ 2)$$

where:
y = Precipitation at the study site (inches)
x = Precipitation at Abbotsford, B.C. Airport (inches)

Therefore, on dates when the weather station at the study site failed, daily precipitation was estimated as 70.7% of that at the Abbotsford Airport (Figure 16; monthly data are compiled in Appendix J, Table J.1, daily data in Table J.2). The annual precipitation at the study site using this method ranged from 40.0 inches in 2005 to 46.2 inches in 2007. The annual precipitation estimates are within 3 inches of the 30-year annual average of 43 inches (70.7% of the Abbotsford Airport 30-year average).

[8] January-April 2006 data were not included due to anomalies in Abbotsford Airport data.

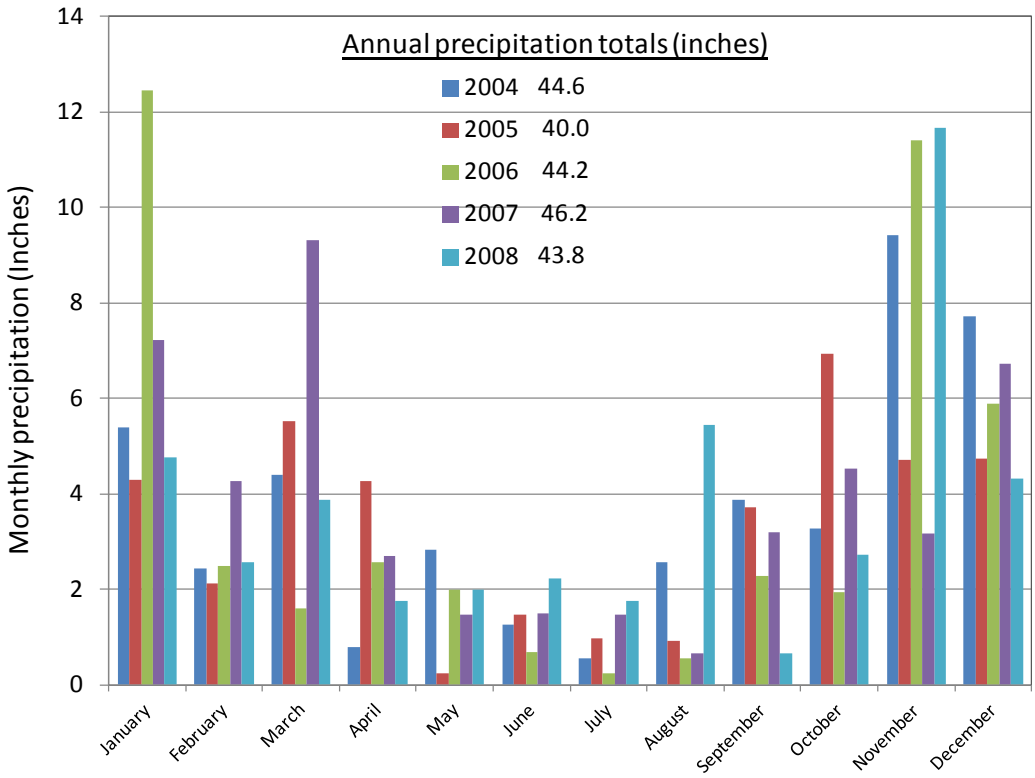

Figure 16. Monthly precipitation at the study site.

Data are from the on-site weather station for 75% of dates. Data for August and September 2004 (before the weather station was installed) and dates when the study site weather station was not operating were estimated as 70.7% of the Abbotsford, B.C. Airport values based on the data regression between the 2 sites.

Air temperature

The average annual daily temperature at the study site was 8.9° to 10.8° C (Table 2). Data from 2 nearby weather stations are included to fill data gaps when the on-site system failed: www.wunderground.com site KWALYNDE1 located between N. Pine Ct. and 19th St. close to the intersection of Guide Meridian and Main St. and the WSU weather station in Lynden (Lat/Long: 49.00176/122.484523).

Table 2. Annual average, minimum, and maximum daily air temperature (ºC) at the study site.
Includes data from KWALYNDE1 and Washington State Univ.-Lynden weather stations when the on-site weather station was not operating. See Appendix J, Table J.3, for daily data.

Year	Average	Minimum	Maximum
2004	10.8	-8.9	25.0
2005	9.6	-6.0	20.3
2006	9.9	-10.2	23.9
2007	9.4	-7.3	25.7
2008	8.9	-9.8	23.0

Soil temperature and soil moisture

Soil temperatures ranged from -5.0° to 29.9 °C (Figure 17, Plate 2). (See Appendix K, Table K.1, for tabular data.) The 5 highest soil temperature measurements occurred in 2006, mostly in the late summer and fall.

Soil moisture measurements ranged from 12.7 to 54.7% (Figure 17, Plate 2). (See Table K.1 for tabular data.) The lowest soil moisture values occurred in the summers of 2005 and 2006. Below a soil moisture level of 20% of dry weight, grass crops commonly go dormant, resulting in little nitrogen uptake (VanWieringen and Harrison, 2009).

Soil temperature and soil moisture tended to be inversely related. When soil temperature was high in the summer, soil moisture tended to be low due to evapotranspiration. Heavy precipitation and low evapotranspiration in the winter kept the soil moist and cool.

On-site soil moisture results early in the study indicated that irrigation applied before the roots became dry could prevent the grass from going dormant in late summer. Therefore, the first application of water occurred earlier in the season each consecutive year in order to maintain grass growth during the dry late summer.

Soil organic matter and soil chemistry

Results for annual soil organic matter and soil chemistry sampling are shown in Appendix K, Table K.2. Soil organic matter ranged from 7.0 to 8.4%. The amount of organic nitrogen available for crops is sometimes calculated at 20 lbs/acre/year for each 1% organic matter, up to a total of 120 lb/acre (NRCS, 2006). The cation exchange capacity of the soil ranged from 19 to 23 milliequivalents/100 g.

Nitrogen and chloride inputs

Nitrogen inputs – manure and inorganic fertilizer

Manure made up the bulk of the total nitrogen mass applied to the field during the study (Figure 18). The timing and quantity of total nitrogen applied as manure (both organic nitrogen and ammonium-nitrogen) and as inorganic fertilizer are shown, by event, on Figure 19 (Plate 2). Inorganic fertilizer was applied only 2 times, once in 2006 and once in 2007 (31 and 48 lb/acre respectively). Atmospheric input was assumed to be 8 lb/acre (Kuipers et al., 2012). The nitrogen contribution from soil organic matter was not measured during the study.

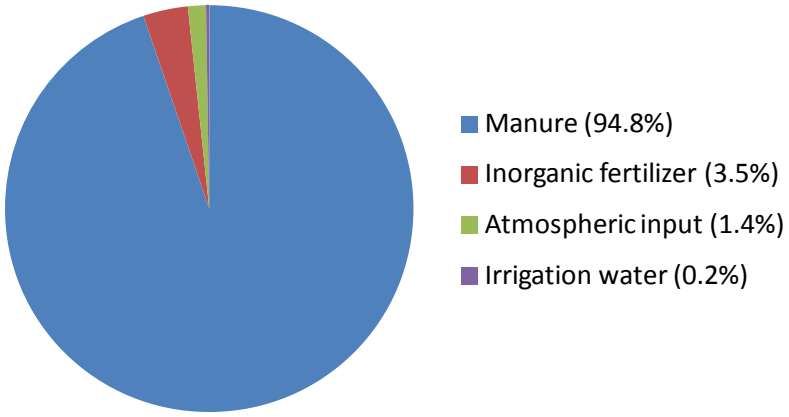

Figure 18. Nitrogen mass inputs to the ground surface by source for 2005 through 2008.
Atmospheric input is from Kuipers et al. (2012).

Manure was first applied each year in the spring. The earliest initial application was on February 18, 2005; the latest on April 27, 2006. The earliest final application for the year occurred on August 31, 2005 and the latest on October 5, 2006. Manure monitoring results are shown in tabular form in Appendix L, Table L.1.

The annual amount of total nitrogen applied by the producer to the field ranged from 394 to 715 lb/acre (Figure 20, Plate 2) with a mean of 548 lb/acre. Between 2005 and 2008, the average nitrogen composition of the applied manure was 47% ammonium-nitrogen and 53% organic nitrogen. The percentage of ammonium was highest in 2005 (67%) and lowest in 2007 (36%) (Figure 19, Plate 2). Nitrate was not measured in manure, because it is typically not a significant component (Beegle et al., 2008).

Most of the nitrogen was applied during the growing season, when there was little if any recharge and uptake by the crop was high. However, depending on the year, between 14 to 25% of nitrogen was applied between October and March (90 to 170 lb/acre/year) when groundwater recharge increases and crop uptake is low.

The average nitrogen content in manure, 15 lb/1,000 gallons, was in the high range compared with manure from 25 Whatcom County dairies with similar solids content (3.5%) reported by Sullivan et al. (1994) (Appendix L, Table L.2). The regression in Sullivan et al. (1994) indicated a total nitrogen content of 10.1 to 12.3 lb/1,000 gallons for manure with 3 to 4% solids. Solids and nitrogen content in manure vary depending on the type of treatment system used prior to field application. Dairies with flush systems, second stage lagoons, and mechanical solids separators typically have lower solids content. However, the dairy where the manure was produced in the current study did not have these systems.

Nitrogen inputs – irrigation water

Annual irrigation water volume totals ranged from 2.5 to 5.7 inches of water (66,000 to 155,000 gallons/acre) as shown in Table 3.

Table 3. Schedule of irrigation water applied.

Year	Number of irrigation events	1st application date	1st application amount (inches/acre)	2nd application date	2nd application amount (inches/acre)	3rd application date	3rd application amount (inches/acre)	Total (inches/ acre)
2005[1]	2	9/15/2005	1.25	10/15/2005	1.25			2.50
2006	2	7/22/2006	1.75	8/22/2006	3.94			5.69
2007	3	7/17/2007	1.96	8/23/2007	1.95	9/12/2007	1.52	5.43
2008	2	7/8/2008	1.95	8/16/2008	2.46			4.41

[1] Dates and amounts are estimates.

Nitrate-N concentrations were measured each year in irrigation water (Figure 21, Plate 2). Ammonium-N and total persulfate nitrogen (TPN) concentrations were analyzed in 2007 and 2008 but not in 2005 and 2006. Because the same source of water was used in 2005 and 2006, and nitrate concentrations were similar in all years, we assumed that the average total nitrogen concentration for 2007 and 2008, 1.3 mg/L-N, was also representative of 2005 and 2006 (Appendix M, Table M.1). The total annual nitrogen mass input from irrigation, 0.8 to 1.8 lb/acre/year, was roughly 0.2% of the total nitrogen applied (Figure 18; Table M.2).

Chloride inputs

The mass of chloride applied during each manure application is shown in Figure 22 (Plate 2) (Appendix L, Table L.2, for data). Chloride was not measured in irrigation water and inorganic fertilizer and is assumed to be negligible in both. The average rate of chloride application to the field in manure was 40 lb/acre/year. The annual total mass of chloride applied was 111 to 205 lb/acre (Figure 23, Plate 2). The application rate for chloride for individual manure applications was correlated with the application rate for ammonium-nitrogen, the most available form of nitrogen in manure (r^2=0.60, n=17).

Nitrogen outputs

Grass crop

The total nitrogen harvested for each grass cutting event is shown in Figure 24 (Plate 2) and was calculated using Equation 3:

$$TN = \left[\frac{CP}{6.25}\right] x\ DM \qquad (Eq.\ 3)$$

where:
TN = Total nitrogen (lb/acre)
CP = Crude protein (%)
DM = Dry matter (lb/acre)
(See Appendix N for results of crude protein, dry matter and total nitrogen removed in the crop.)

Figure 25 (Plate 2) shows the annual totals of nitrogen harvested. The highest annual total nitrogen yield occurred in 2007 (457 lb/acre), when the total nitrogen applied was 434 lb/acre. The lowest annual nitrogen uptake, 393 lb/acre, occurred in 2008, the year with the highest amount of nitrogen applied (715 lb/acre).

The estimate for 2005 nitrogen harvested includes results for the last grass crop of the year (102 lb/acre), which was not actually removed from the field due to inclement weather. This unrealized harvest was included to represent the uptake of nitrogen for the year, even though it was not removed from the field. This biased the 2005 crop uptake estimate high, because some of the grass that was not removed decomposed and became available for leaching.

Nitrogen residual – soil nitrate

Soil nitrate results are shown in Figure 26 (Plate 2; data in Appendix K, Table K.1). The range for soil nitrate concentrations (August through November) was 5.5 to 60 mg/kg.

Equation 4 was used to convert mg/kg dry weight of soil nitrate to lb/acre. The bulk density value for the Hale silt loam soil (0- to 10-inch depth) at the site is 1.13, assuming the average organic matter content at the site = 7.5%. Therefore, the conversion factor from mg/kg nitrate dry weight to lb/acre ($\rho_b\ x\ 2.791$) is 3.1. This is somewhat lower than the suggested conversion for western Washington, 3.5, in Sullivan and Cogger (2003).

$$N = C_{Soil\ N}\ x\ \rho_b\ x\ 2.79 \qquad (Eq.\ 4)$$

where:
N = Soil nitrate concentration (lb/acre)
$C_{Soil\ N}$ = Soil nitrate concentration, dry weight (mg/kg DW)
ρ_b = Soil dry bulk density (g/cm^3)
2.791 = Unit conversion constant [to convert mg nitrate-N/kg (dry weight) to lbs nitrate /acre-ft]

The data results indicate:

- The range of soil nitrate concentrations observed during the September to October period, when post-harvest soil nitrate samples are usually collected (Plate 2, Figure 26, green boxes), was 11.5 to 43 mg/kg (36 to 133 lb/acre).
- The range of concentrations observed for the 10 samples that were collected on a date that met the guidance provided by Sullivan and Cogger (as soon as possible after the last manure application and before 5 inches of precipitation starting September 1; see Table 4) was 15.5 to 43 mg/kg (48 to 132 lbs/acre).
- If sample dates in November are included in the post-harvest period, the soil nitrate concentration observed ranged from 5.5 to 60 mg/kg (17 to 186 lbs/acre).
- The highest value observed, 60 mg/kg (186 lb/acre) on November 8, 2006, was outside of the recommended sampling period.
- Soil nitrate concentrations were highly variable. There was typically a 2-fold difference between the maximum and minimum weekly soil nitrate concentrations in the fall season each year (September 1 through October 31), or up to 24 mg/kg (74 lb/acre). Temperature, precipitation, spatial heterogeneity, timing of the last manure application, and other factors influence these changes.

Table 4. Post-harvest soil nitrate results that met the timing protocols recommended in Cogger and Sullivan (2003).

Date	Soil nitrate concentration (mg/kg)	Soil nitrate mass (lb/acre)
9/9/2004	43.0	133
9/17/2004	28.5	88
10/1/2004	19.0	59
10/4/2005	15.5[1]	48
10/11/2005	16.0[1]	50
10/9/2007	16.9	52
10/16/2007	18.8	58
10/17/2008	26.9	83
10/24/2008	30.8	95
10/31/2008	29.9	93

[1] Last manure application occurred after the last harvest.

Groundwater conditions

Hydrogeologic conditions

Aquifer properties

Grain size distribution

Split-spoon soil samples from monitoring well borings analyzed for grain size distribution were used to classify soil samples according to ASTM Method 2487-92 (ASTM, 1994). Table 5 lists the values for effective grain size (d_{10}), uniformity coefficient (C_u), and coefficient of curvature (C_c). These values were calculated using particle size distribution curves and were used to classify soils. Soil classification results are shown in cross-section in Figure 27 (Plate 1). See Appendix O for particle size distribution curves.

The effective grain size, d_{10}, is used here to qualitatively compare the potential rate of leaching and the potential for denitrification. The d_{10} represents the sieve diameter through which only the smallest 10% of the particles pass. The lower the d_{10} value, the larger the portion of fine-grained material in the sample. Slow percolation of liquid through fine-grained material allows for bacterial or chemical processes that use up oxygen and enhance denitrification potential if there is a sufficient electron source such as organic carbon.

The uppermost sediments varied among fine-grained classifications of clay or silt with sand and sand with silt and clay. At side-by-side borings, AKG725 and AKG726, samples from 7.5 to 25 feet below ground surface (BGS) contained little silt or clay and were categorized as well graded sand. Samples below 7.5 feet in the other wells had varying amounts of fine-grained material. But samples from all wells indicated more rapid movement of water vertically and horizontally below 7.5 feet than at shallower depths. The deepest sample was collected from the top of the confining layer that forms the base of the aquifer at 40 feet (AKG726).

Table 5. Particle size distributions for split spoon soil samples collected during installation of monitoring wells.

Well	Depth (feet BGS)	Soil class[1]	Description	d_{10} (mm)[2]	C_u[3]	C_c[4]	D_{60}	D_{10}	D_{30}
AKG-721	2.5	SM or SC	Sand with silt or clay	<0.075	166.7	4.8	0.500	0.003	0.085
AKG-721	5.0	ML with sand, or CL with sand	Silt or clay with sand	<0.001					
AKG-721	10.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.078	3.2	1.3	0.270	0.085	0.170
AKG-722	5.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.122	2.0	2.3	0.240	0.122	0.260
AKG-722	10.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.115	2.4	0.9	0.280	0.115	0.170
AKG-723	2.5	CL or ML with sand, or CL-ML with sand	Clay or silt with sand or silty clay with sand	<0.001					
AKG-723	10.0	SP-SM or SP-SC	Poorly graded sand with silt or clay	0.087	4.4	2.2	0.380	0.087	0.270
AKG-724	7.5	SP	Poorly graded sand	0.169	4.4	1.4	0.750	0.169	0.420
AKG-725	2.5	CL or ML with sand, or CL-ML with sand	Clay or silt with sand or silty clay with sand	<0.075					
AKG-725	7.5	SW	Well graded sand	0.096	2.7	1.2	0.260	0.096	0.170
AKG-726	15.0	SW	Well graded sand	0.139	1.7	2.0	0.230	0.139	0.250
AKG-726	25.0	SW	Well graded sand	0.109	2.8	1.2	0.300	0.109	0.200
AKG-726	40.0	CL or ML with sand, or CL-ML with sand	Clay or silt with sand or silty clay with sand	<0.0013					
AKG-727	2.5	SM or SC	Sand with silt or clay	<0.0013	120.8	15.5	0.145	0.001	0.052
AKG-727	10.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.087	3.2	1.3	0.260	0.082	0.165

[1] Plasticity index and liquid limit were not determined; therefore, silt and clay could not be distinguished. SW=Well-graded sand, SP=Poorly-graded sand, SM=Silty sand, SC=Clayey sand, CL=Clay, ML=Silt.

[2] Effective grain size: Particle size diameter through which 10% of sample particles pass on cumulative particle size distribution curve.

[3] $C_{u:}$ D_{60}/D_{10} (Coefficient of Uniformity – if 1-3, then well graded; if greater than 3, then poorly graded).

[4] $C_{C:}$ $(D_{30})^2/(D_{10} \times D_{60})$ (Coefficient of curvature measures the shape of the particle size curve indicating gradation).

Horizontal hydraulic conductivity

The Bradbury and Rothschild (1985) method was used to estimate horizontal hydraulic conductivity (K_H), a measure of the permeability of the aquifer sediments (see Appendix P for details). Hydraulic conductivity is used to estimate the velocity of groundwater flow.

Specific capacity results and estimated K_H values for 3 on-site monitoring wells are shown in Table 6. Two of the wells (AKG725 and AKG726) are only 3 feet apart but are screened at different depth intervals. Monitoring wells AKG725 and AKG723 are screened from 6 to 13 feet below ground surface (BGS); AKG726 from 25 to 35 feet BGS. The average K_H value at the site was approximately 53 feet/day.

Table 6. Horizontal hydraulic conductivity (K_H) estimates based on specific capacity. *Bradbury and Rothschild, 1985.*

Well I.D.	Static water level (feet)	Pumping water level (feet)	Saturated screen length (feet)	Assumed storage coefficient	Aquifer thickness (feet)	K_H (feet/sec)	K_H (feet/day)
AKG726	8.77	8.98	10.0	0.20	35	7.95E-04	69
AKG725	8.72	8.78	4.3	0.20	35	6.07E-04	53
AKG723	7.98	8.06	4.7	0.20	35	4.17E-04	36

K_H: Horizontal hydraulic conductivity (rate of flow through a material over time at a unit gradient).

Groundwater occurrence and movement

Groundwater elevations

Hydrographs of water level elevations are shown in Figure 28 (Plate 3). The highest water levels occurred in the winter (December through March); the lowest in the fall (September through October). (See Appendix Q, Table Q.1, for data in tabular form.) On January 10 to 11, 2006, monitoring wells AKG722 and AKG727 were submerged and could not be monitored. Subsequent water quality and water level data did not indicate leakage from surface water to the well screen.

Depth to water

Depth to water from the top of the casing in the monitoring wells ranged from 0 to 11.4 feet (Figure 29, Plate 3). (See Table Q.2 for tabular data.) The shallowest water table values (0 to 5.2 feet BGS) occurred in winter months, coincident with the period of highest potential for nitrate leaching and the lowest potential for crop uptake of nutrients (December through March). The deepest annual water table depths, 10.4 to 11.4 feet, usually occurred in October.

The annual range of depth-to-water measurements in individual wells between highest and lowest depths was 4.5 to 10.1 feet/year. The mean annual difference between high and low water table depths was 7 feet.

Groundwater flow direction

Water level contours typical for fall low conditions (October 1, 2007), high winter conditions (December 28, 2004), and middle-range spring and summer conditions (March 2, 2005 and June 26, 2006) are shown in Figure 30 (Plate 3). The groundwater flow direction was consistently south-southwest toward the Nooksack River.

Hydraulic gradient

The horizontal hydraulic gradient ($i_{H;}$ dimensionless), or slope of the water table, was calculated using Equation 5:

$$i_H = \frac{dh}{dl} \qquad \textit{(Eq. 5)}$$

where:
dh = change in hydraulic head between 2 points (feet)
dl = lateral distance between 2 points (feet)

Hydraulic gradients for the contours in Figure 30 ranged from 0.0027 to 0.0051 with an average of 0.0037 (Table 7). Hydraulic gradients tended to be lowest in the late summer to fall season and highest during the winter.

Table 7. Horizontal hydraulic gradient estimates at the study site on 4 representative dates.
See Figure 30 (Plate 3).

Date	Horizontal hydraulic gradient (i_H)
12/28/2004	0.0051
3/2/2005	0.0037
6/26/2006	0.0033
10/1/2007	0.0027

Positive vertical hydraulic gradients were measured at the side-by-side shallow (13 feet deep) and deep (38 feet deep) wells, AKG725 and AKG726, indicating a downward hydraulic potential throughout the study period (Figure 31, Plate 3). An increasingly positive trend in the vertical hydraulic gradient value over the study period indicates that water and dissolved constituents moved downward more quickly over time. The mean vertical hydraulic gradient was 0.0047 (Standard deviation = 0.001, n=43).

Groundwater flow velocity

The average horizontal velocity of groundwater flow was estimated using a variation of Darcy's Law:

$$v = \frac{-K_H(\frac{dh}{dl})}{n_e} \qquad \textit{(Eq. 6)}$$

where:
v = Average linear groundwater velocity (feet/day)

K_H = Horizontal hydraulic conductivity (feet/day)
dh/dl = Horizontal hydraulic gradient (dimensionless)
n_e = Effective porosity (ratio of the volume of interconnected voids/volume of material that is capable of transmitting fluid)

We used the K_H value for well AKG725 (Table 8), because it is in the middle of the field and probably most representative of the shallow water table at the site. The annual range of horizontal hydraulic gradient was used for dh/dl.

The range of groundwater velocity estimates was 0.57 to 1.08 feet/day, or 208 to 394 feet/year, with a mean value of 0.78 foot/day, or 284 feet/year (Table 8).

Table 8. Estimates of groundwater velocity at the study site using the minimum, average, and maximum hydraulic gradients for dates shown in Table 7.
See Figure 30 (Plate 3).

Hydraulic gradient	Hydraulic conductivity (K_H)[1] (feet/day)	Hydraulic gradient (feet/feet)	Effective porosity[2]	Velocity (feet/day)
Minimum	53	0.0027	0.25	0.57
Average	53	0.0037	0.25	0.78
Maximum	53	0.0051	0.25	1.08

[1] From Table 6.
[2] Representative value for glacial outwash aquifers.

Recharge

Recharge was estimated using the water balance method shown in Equation 7 (Healy and Scanlon, 2010; Malekani, 2012):

$$R = P - PET \qquad (Eq.\ 7)$$

where:
R = Monthly recharge (inches)
P = Monthly precipitation (inches)
PET = Monthly potential evapotranspiration (inches)

Monthly potential evapotranspiration data were drawn from estimates developed by Environment Canada for the Abbotsford Airport climate station. These estimates were calculated using a modification of the Penman Monteith Equation for a grass crop (www.farmwest.com/node/930). During months when PET exceeded precipitation (generally May through August), recharge was assigned a value of 0 inches.

Annual recharge estimates ranged from 23.5 to 29.4 inches and are shown on Figure 31 (see Appendix J, Tables J.4 and J.5, for tabular data). Recharge comprised an average of 62% of annual precipitation, when annual precipitation was calculated from September 1 to August 31.

This calculation period is similar to the water year used in hydrologic budgeting (October 1 to September 30) but shifted back one month to fit climate patterns for the area and nitrate leaching potential. An average of 65% of the annual recharge occurred from October through March. This is similar to previous reports (Kuipers et al., 2012).

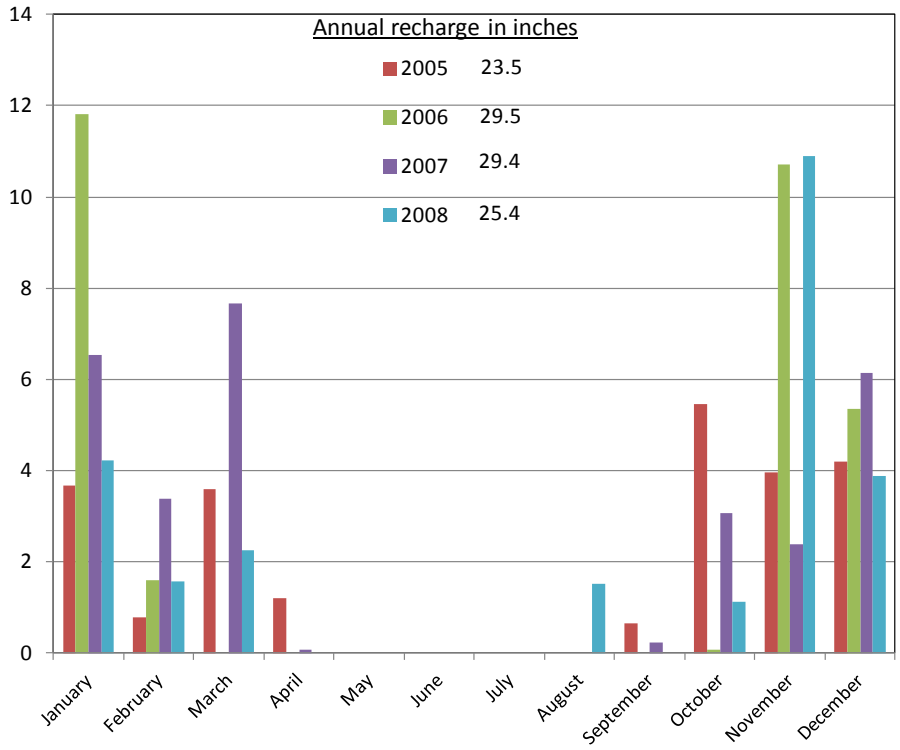

Figure 32. Estimated monthly recharge for the study site in inches.

Most of the focus on nitrate loss to groundwater is on the fall/early winter season; however, recharge and associated leaching continues to occur in late winter/early spring when precipitation exceeds evapotranspiration (Zebarth, 2013; Kuipers et al., 2012; Chesnaux and Allen, 2007).

Groundwater quality conditions

Time series results for groundwater quality are shown in Plates 4 and 5. Results are shown for pH, dissolved oxygen, conductivity, nitrate-N, chloride, organic carbon, total dissolved solids, and total phosphorus. Total nitrogen concentrations were similar to nitrate-N and are not shown graphically. Samples collected at well AKG727 between February 7 and July 14, 2005 were rejected as described in the *Quality Assurance* section and are not included in the results summary. See Appendix R, Table R.1, for monitoring well groundwater quality data in tabular form. Table R.2 summarizes results from upgradient private wells.

pH

pH affects the oxidation/reduction state of ammonia in groundwater. When pH is below 8, most of the ammonia is in the ammonium form (NH_4^+). This is the case in most of western Washington groundwater and surface water. pH also affects the rate of bacterial conversions of ammonia to nitrate (nitrification) and nitrate to nitrogen gas (denitrification) (Buss et al., 2004 and 2005; Coyne, 2008). (See *Nitrogen Cycle* section above.)

Results for groundwater pH shown in Figure 33 (Plate 4) were all below 8, indicating that ammonium was the predominant form of ammonia in groundwater at the study site. pH values ranged from 5.1 to 6.0 in all monitoring wells except AKG724 and AKG726 and did not vary significantly seasonally. The lowest pH values occurred in well AKG724, where values were consistently below 5.0. The highest pH occurred in the deep well, AKG726, with values typically around 6.5.

Dissolved oxygen (DO)

Results for DO are shown in Figure 34 (Plate 4). The DO concentration has a major influence on the potential for denitrification to occur, as well as the oxidation state of nitrogen and phosphorus. When the DO concentration is less than 1 to 2 mg/L and organic carbon (or other electron donor) is in sufficient supply, bacteria convert nitrate to nitrogen gas (Buss et al., 2005; Bates and Spalding, 1998). Denitrification can also occur at microsites in soil and aquifer materials, where the bulk DO is higher than 2 mg/L, but is generally more significant where the bulk water is also low in DO (U.S. EPA, 2013).

DO concentrations were consistently above 2 mg/L in samples from wells AKG721 and AKG725 in the northwest part of the field. Most measurements in these wells were in the range of 6 to 10 mg/L, far above the level where denitrification occurs. In the other shallow monitoring wells, DO concentrations followed a seasonal pattern, with oxygen decreasing during the late summer, sometimes below 2 mg/L, probably due to elevated bacterial activity.

In the winter, shallow groundwater DO concentrations were rapidly replenished with oxygen-rich recharge from precipitation. The highest DO concentrations in most wells occurred in January and February following a water table rise of up to 7 feet within several weeks. Monitoring wells on the east side of the site followed this pattern most closely (e.g., AKG722).

DO concentrations in the deep well, AKG726, and the upgradient private wells were consistently at or close to 0.0 mg/L (anoxic). Occasional observations above 0.2 mg/L in AKG726 were not made using the standard sealed flow cell, because it was unavailable. Instead purge water was directed into the bottom of a 5-gallon bucket with the DO probe also at the bottom. The slightly higher values are probably an artifact of measuring in an open container.

Specific conductance and total dissolved solids (TDS)

Specific conductance and TDS results (Figures 35 and 36, Plate 4) followed similar patterns throughout the study. Both parameters generally increased in the fall-winter of 2004-2005,

declined slightly, and peaked again in summer 2005. During the following 3 years, both parameters gradually declined until December 2008, when 3 of the shallow wells (AKG722, AKG723, and AKG725) showed substantial increases.

All conductivity and TDS results were below the secondary maximum contaminant levels (MCLs) for drinking water, 700 umhos/cm for conductivity and 500 mg/L for TDS (Chapter 246-290 WAC).

Chloride

Chloride concentrations are shown in Figure 37 (Plate 5) and ranged from 4.4 to 30.6 mg/L with the highest concentrations in 2004 and 2005. All results were below the secondary MCLs for drinking water of 250 mg/L (Chapter 246-290 WAC).

Patterns observed in chloride concentrations were similar to those observed for specific conductance and TDS. Concentrations of chloride were initially higher in the shallow well, AKG725, than in the nearby deep well, AKG726. Like most of the shallow wells in the study, chloride decreased at AKG725 for the first 3 years of the study until late 2008. Occasional increases in chloride at AKG725 corresponded with manure applications followed by heavy precipitation (see Figures 19 and 37, Appendix J). A particularly large increase in chloride occurred at AKG725 on August 16, 2005, when chloride reached 30 mg/L, indicating leachate reaching the water table even in the summer.

Chloride decreased slightly over time in the deep well, AKG726. Chloride concentrations in upgradient domestic water supply wells ranged from 16 to 18 mg/L.

Dissolved organic carbon (DOC)

DOC results are shown in Figure 38 (Plate 5). All organic carbon data collected before February 5, 2005 represent total organic carbon (no filtering). Samples collected on February 5, 2005 and afterward were filtered in the field and represent DOC except samples from AKG726. Samples from AKG726 were not filtered and represent total organic carbon, because the in-line filtering system was not equipped for the submersible pump needed for the deeper well.

AKG722 consistently had the highest DOC concentrations, with seasonal fluctuations that mimicked the water table elevations with about one month lag time (Figure 39). The maximum DOC observed was 9.6 mg/L at AKG722 on February 27, 2008. DOC in the other shallow wells sometimes fluctuated with the water table elevation but to a lesser extent than at AKG722.

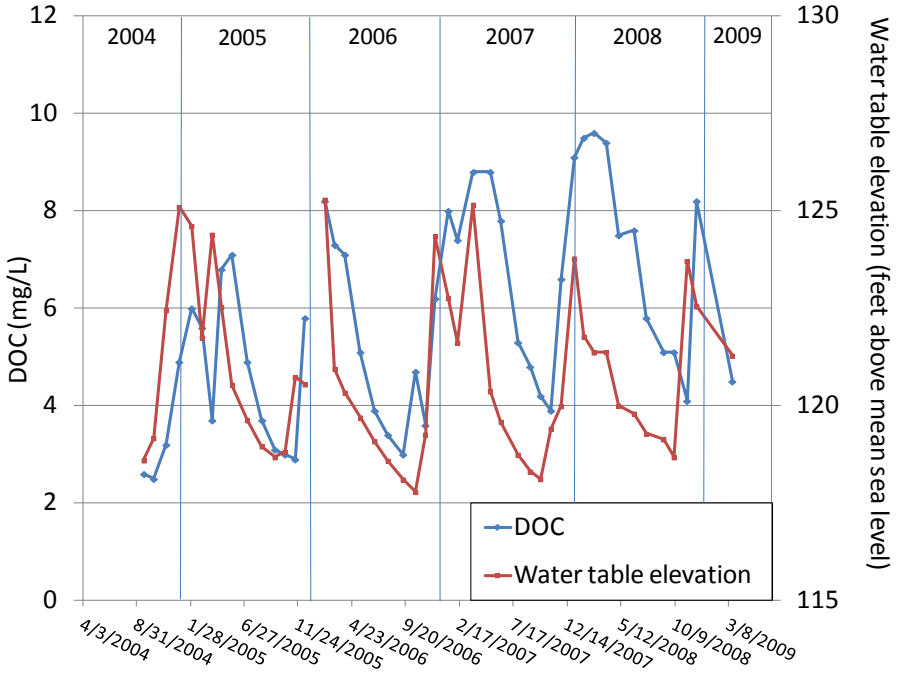

Figure 39. Dissolved organic carbon concentrations and water level elevations in monitoring well AKG722.

Nitrate

Nitrate results are shown in Figure 40 (Plate 5). Nitrogen was predominantly in the nitrate form in all wells except the deep, mostly anoxic well, AKG726. Concentrations of nitrate in shallow groundwater were well above 10 mg/L-N at the beginning of the study, except at AKG722. During the winter months of 2004 to 2005, nitrate concentrations increased in all wells to the highest levels observed during the study. The concentration at AKG722, 45 mg/L, was more than 4 times higher than the Washington State groundwater standard and federal MCL for drinking water.

During the second and third years of the study, 2006 to 2007, nitrate concentrations in the shallow wells decreased at about the same rate as declines in chloride concentration. Within the general decline, nitrate fluctuated somewhat. During the last few months of the study, December 2008 through March 2009, nitrate increased substantially in 4 out of 6 shallow wells, one reaching 20 mg/L-N (AKG725). Chloride followed a similar pattern.

In the anoxic deep well, AKG726, nitrate concentrations ranged from below the detection limit of 0.01 mg/L-N to 0.333 mg/L-N.

Concentrations of total persulfate nitrogen (TPN) were very similar to nitrate-N in shallow monitoring wells, indicating that little organic nitrogen was present.

Unfiltered nitrate concentrations in upgradient domestic water supply wells screened at 29 to 33 feet depth were 0.014 to 0.021 mg/L-N.

Ammonium

Results for ammonium-N are shown in Figure 41. Ammonium is typically attenuated in the soil, because positively charged ammonium ions ($NH4^+$) adhere to negatively charged soil particles (Buss, 2004). Because the pH was below 6 in the shallow monitoring wells and below 7 in the deep well, almost all of the ammonia is in the ammonium form. Therefore, for groundwater we will refer only to ammonium-nitrogen in this report.

Ammonium concentrations in shallow groundwater were below the detection limit of 0.01 mg/L on 37 out of 46 dates. The highest number of detections occurred on October 18 to 19, 2004, when the range of concentrations in 6 wells was 0.012 to 0.018 mg/L-N. Ammonium was the main nitrogen species found in samples from the deep well, AKG726, with concentrations ranging from less than 0.170 to 0.248 mg/L-N. Ammonium concentrations in upgradient domestic wells ranged from 0.194 to 0.255 mg/L-N.

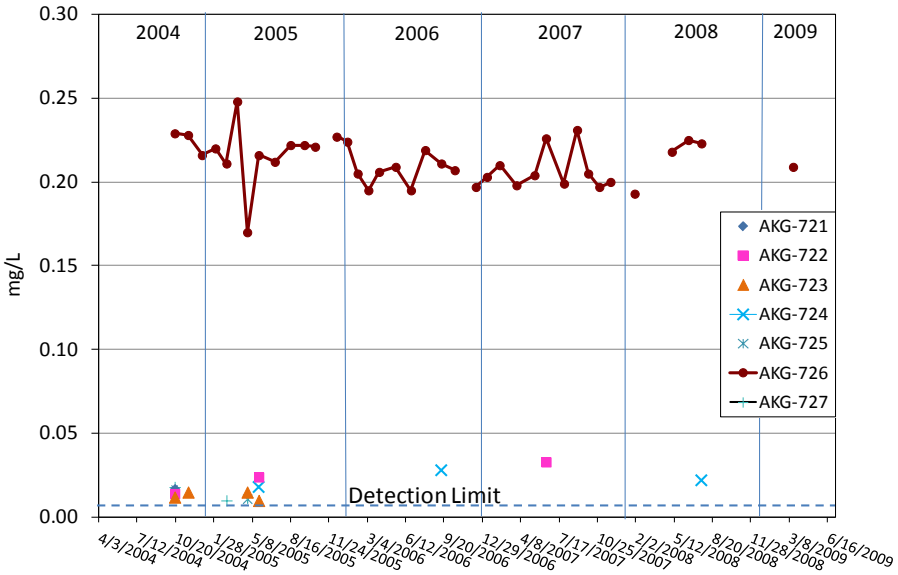

Figure 41. Ammonium-N results in monitoring wells.

Total persulfate nitrogen (TPN)

TPN concentrations were very similar to nitrate and are listed in Appendix R, Table R.1.

Total dissolved phosphorus (TDP)

TDP concentrations were monitored only from 2004 to 2006, because TDP was not the main focus of the study. Samples from AKG726 were not filtered and therefore represent total phosphorus (TP).

Results for phosphorus in groundwater are shown in Figure 42 (Plate 5). Shallow groundwater TDP ranged from 0.0005 to 0.0129 mg/L. Results for TP in the deeper anoxic groundwater were somewhat higher (0.113 to 0.264 mg/L) but are potentially biased high, because the samples were not filtered.

Discussion

The study results are discussed below from the perspective of nitrate movement, groundwater quality and aquifer characteristics. Manure, soil, and crop results are used to interpret groundwater results. For a detailed interpretation of manure, soil, and crop results, see VanWieringen and Harrison (2009).

Factors that influence nitrate conditions in groundwater and soil

The groundwater and soil nitrate conditions observed during the study were the result of a complex interaction of many environmental and farm management factors. These factors, and their influence on nitrate dynamics and fate, are discussed individually below.

Environmental factors

Hydrogeologic conditions

The Abbotsford-Sumas Aquifer, of which the Sumas-Blaine Aquifer (SBA) is a part, is not homogeneous. Grain size and depth to water vary spatially. These characteristics can play an important role in the vulnerability of different portions of the SBA to nitrogen leaching. Properties of the vadose zone and aquifer at the study site are discussed below and compared with information from other parts of the aquifer.

Grain size distribution

Movement of water and dissolved nitrate to the water table and along the groundwater flow path are affected by the texture of the substrate. Water penetrates more slowly through finer materials, which can lead to higher winter surface runoff and reduced nitrogen loading to the aquifer.

Finer soils over the central and western parts of the SBA tend to become saturated in the winter due to lower infiltration capacities, flat topography, and rapidly rising water tables. Recharge water percolating to the water table is enriched in oxygen. During the summer, when recharge is limited, oxygen can become depleted by bacterial consumption in finer soils, and denitrification is more likely. Slower velocity in fine-grained materials also allows more time for bacterial consumption of oxygen than in coarser materials.

Denitrification, which removes nitrate from the subsurface, is less likely in the coarser, well-drained soils and aquifer material in other parts of the SBA. There is more interconnected pore space in coarse-grained material, infiltration rates are faster, and replenishment of oxygen exceeds oxygen consumption by microorganisms. Paul and Zebarth (1997) found that denitrification accounted for only 17% of annual nitrogen loss from medium to coarse soils in south-coastal British Columbia following dairy manure application. The remaining 83% of soil nitrate was presumed to leach to groundwater.

The Natural Resources Conservation Service (NRCS, 2006) estimates nitrate loss of 15% in western Washington soils, where the depth to water is greater than 2 feet below the soil surface. Van Es et al. (2006) found that the nitrate concentration in drainage outflow beneath manured fields with loamy sand (coarse-grained) was on average twice what emerged from beneath loamy clay (fine-grained). de Ruijter et al. (2007) likewise found consistently higher groundwater nitrate concentrations in coarse-grained materials than in fine-grained materials at 34 farms studied in the Netherlands.

Figure 43 (Plate 1) compares the effective grain size results from well borings at the study site with locations shown on Figure 44 (Plate 1). Samples from 5-foot to 30-foot depths tend to grade coarser from west to east, with the Abbotsford samples having 10 times higher effective grain size values, at 16 and 26 feet, than the other sites. The Abbotsford samples indicate much coarser material in the northeastern part of the SBA than found to the west and southwest.

Particle size analyses from monitoring-well core samples indicated relatively fine-grained material at the screened depth, where sample water enters the well, in 4 out of 6 monitoring wells (sand with silt or clay) and coarser sand in 2 of the wells (AKG724 and AKG725). We saw evidence of denitrification in the wells with finer-grained material (except AKG721) as well as one well in coarser-grained material (AKG724). Core samples from shallower depths (2.5 feet) were extremely fine-grained in most samples (d_{10} less than 0.001 at 3 out of 4 bore holes).

These patterns suggest that nitrate losses due to denitrification are probably higher at the study site than in coarse-grained parts of the SBA, and especially higher than in very coarse-grained eastern parts of the aquifer. Rates of nitrate infiltration may also be lower at the study site than in coarser areas of the SBA.

Hydraulic conductivity and groundwater velocity

Hydraulic conductivity (K_H) estimates for the study site, 36 to 69 feet/day (average 53 feet/day), were lower than 84% of wells in the SBA analyzed by Cox and Kahle (1999). The lower K_H indicates slower flow than most locations where information is available in the aquifer. Lower groundwater velocities can result in a higher denitrification potential, likely making the study site somewhat less vulnerable to nitrate contamination by leaching than areas with higher K_H values.

After entering the aquifer, water and solutes at the study site move mainly horizontally in a southerly direction below the site. The average groundwater flow velocity at the site, 0.78 foot/day, is lower than the average value for the Abbotsford-Sumas Aquifer reported by Cox and Kahle (1999) of 2 feet/day. Lower velocity may lead to lower dispersion in the aquifer and slower decrease in nitrate concentration along the groundwater flow path.

Moisture content and preferential flow

Flow of water through the vadose zone to the water table occurs through both saturated and unsaturated flow. In unsaturated soils, as water content increases, the hydraulic conductivity and soil pore-water drainage generally increase (Wierenga, 1995). The change in hydraulic conductivity over the range of unsaturated conditions is less pronounced for fine-grained soils like those at the study site than for coarse soils in some locations overlying the SBA. In general,

however, more and faster downward flow of water would be expected from the vadose zone to the water table with increasing soil moisture during the wet season (October through March).

Flow of water and dissolved constituents, such as nitrate, below the surface via cracks and openings as small as a few microns in diameter, can be rapid, inherently variable over a field, and difficult to track (Nimmo, 2013; Osnoy et al., 2005; Selker, 1999). Downward movement of percolating water prevents crop uptake or other processes that might otherwise remove nitrate or alter the chemistry of the water.

We assumed that most of the nitrate lost to leaching below the field reached the water table during the high-precipitation period (October through March). However, downward movement probably also occurred during the rest of the year to a lesser degree, especially following irrigation events.

Depth to water

Shallow groundwater depth observed during this study provides a short transport route for nitrate and dissolved constituents to groundwater unless the percolating water is redirected via tile drains to a surface water body. Tile drains are not present in the study field but are common in the low-lying area overlying the SBA. The winter water table at the study site was typically within 0 to 4 feet of the surface, intersecting the root zone of the crop and resulting in direct dissolution of nitrate into groundwater.

Air temperature

Crop growth and nitrogen uptake generally increase with warmer temperatures, leaving less excess nitrogen in the soil at the end of the growing season. VanWieringen and Harrison (2009) evaluated the influence of temperature on crop removal during the study using 3 methods that estimated growing degree units (GDUs). All 3 methods indicated that 2008 was significantly cooler than the other years and that most of the year-to-year variation in grass yield was due to temperature.

The Griffith and Thompson (1996) method used by VanWieringen and Harrison (2009) for estimating GDUs appears to best fit the study location and crop and is shown in Equation 8.

$$GDU = [(T_{Max} + T_{Min})/2] - 32 \qquad \text{(Eq. 8)}$$

where:
GDU = Monthly growing degree units (°F)
T_{Max} = Maximum monthly temperature (°F)
T_{Min} = Minimum monthly temperature (°F)

The annual total GDUs for January through October are shown in Figure 45 (Plate 6). The highest year for thermal input to the grass crop, 2007, coincided with the highest crop nitrogen removal (Figure 46, Plate 6), one of the lowest annual nitrogen application rates (Figures 47, Plate 6), and groundwater nitrate values fluctuating around 10 mg/L-N (Figure 40, Plate 5). Likewise the year with the lowest annual GDU total, 2008, coincided with the lowest crop

nitrogen removal. The lower crop removal and higher excess nitrogen in 2008 resulted in a return to shallow groundwater nitrate concentrations above 10 mg/L-N in most wells.

Influence of upgradient conditions on groundwater quality

Six out of 7 monitoring wells used for this study were constructed in the uppermost portion of the SBA. The open intervals of the wells intersected the water table, where the most recently recharged water was located. This allowed us to monitor groundwater quality responses to nitrogen cycling and loading in the near vicinity of each shallow well.

A shallow well specifically designed to represent the ambient groundwater quality at the upgradient boundary of the field was not installed for the study. Groundwater samples collected from the upgradient domestic wells are not considered representative of the *shallow* background condition at the far northern (upgradient) end of the study field, because of the position of their deeper open intervals 29 to 33 feet below ground surface (Figure 11).

Although the water quality of the shallow groundwater entering the upgradient boundary of the site was not directly monitored, it is unlikely that conditions beneath properties north of the study area had a significant influence on sample results during the study period. Our reasoning for this conclusion includes:

- The upgradient location, where a groundwater sample from a given depth first entered an aquifer, can be approximated using the following set of equations (Harter, 2013):

$$D_{Recharge} = \frac{R}{n_e} \qquad (Eq. 9)$$

where:
$D_{Recharge}$ = the maximum depth of recharge movement into the aquifer (feet)
R = the total amount of recharge during the period of interest (feet)
n_e = the effective porosity of the aquifer sediments (dimensionless)

then:

$$GW_{Age} = \frac{R}{R_{Annual}} \qquad (Eq. 10)$$

where:
GW_{Age} = the maximum age of a groundwater sample of a given depth (year)
R_{Annual} = the total annual recharge rate (feet/year)

then:

$$TD_{GW} = GW_{Age}\ x\ V_{GW} \qquad (Eq. 11)$$

where:
TD_{GW} = the approximate distance a groundwater sample collected from a given depth traveled in the aquifer from its point of entry as recharge (feet)
V_{GW} = the average linear groundwater velocity (feet/year)

An estimate of the maximum age of a groundwater sample collected from a well can be made by setting the $D_{Recharge}$ term equal to the maximum depth of the well's open interval below

the water table and rearranging Equation 10 to solve for (*R*). Once determined, the maximum age of the sample (i.e., the age of the water drawn into the lowest point of the open interval) can then be integrated with the linear groundwater flow velocity at the site to approximate the greatest distance the sample traveled before arriving at the well.

The lowermost portions of the open intervals of the shallow wells at the site were typically no greater than 9 feet below the water table. Applying site-specific values to the equations above (n_e = 0.25; V_{GW} = 285 feet/year; R_{Annual} = 2.25 feet; $D_{Recharge}$ = 9 feet), the approximate maximum travel distance (from the original point of entry as recharge) for groundwater samples collected from these wells is 285 feet. Water entering the well through the uppermost portions of the open interval is likely more recently recharged and has traveled a shorter distance through the aquifer.

Although this method greatly simplifies groundwater transport through an aquifer and into a well, the analysis suggests that samples collected from the shallow monitoring wells largely originated within the study field itself. Most of the monitoring wells are likely too shallow to capture any significant fraction of water originating upgradient of the study field. The mid-field and southernmost shallow wells (AKG722, AKG723, AKG724, and AKG725) were all located more than 500 feet downgradient from the upgradient edge of the field (Figure 48). This suggests that by the time groundwater travels from upgradient offsite sources to these wells, it has moved too deep into the aquifer to intersect the shallow well screens.

- Groundwater samples from the on-site well screened at the base of the aquifer (AKG726) are not considered representative of current management practices at the study site. Because the open interval for this well (28 to 38 feet) is so deep, samples from AKG726 probably represent water that entered the aquifer far upgradient of the study field (~1,200 feet, based on Equations 9-11). The effects of denitrification near the base of the aquifer appear to have removed most of the nitrate that was initially present in the groundwater when it first arrived at the water table.

- A 3.7-acre residence lies immediately upgradient (north) of the study site. The potential upper-range nitrogen input from the on-site sewage system at the residence was 36 lb nitrogen/year [9 lb nitrogen/person/year according to U.S. EPA (2002) times 4 residents living in the house full time] and potentially approximately 85 lb nitrogen/acre for lawn care on 2 acres (170 lb nitrogen total).

  The total estimated annual nitrogen loading to 3.7 acres would therefore be 206 lb nitrogen/ 3.7 acres or 56 lb total nitrogen/acre. This is 8 to 13% of the amount of total nitrogen applied to the study field between 2005 and 2008. This suggests that the water quality impact on the closest shallow monitoring well to this property (AKG721) was relatively insignificant in comparison to manure loading at the study site.

- The property farther upgradient of the residence was cultivated during the study and probably received manure at a rate similar to rates observed at the study field. However, the field lies 390 feet north of the upgradient edge of the study field, beyond the distance that samples from the shallow monitoring wells would likely be affected.

On the basis of the evidence above, the groundwater quality results presented in this report are assumed to largely represent groundwater quality responses to nitrogen loading and management occurring directly on the surface of the study field.

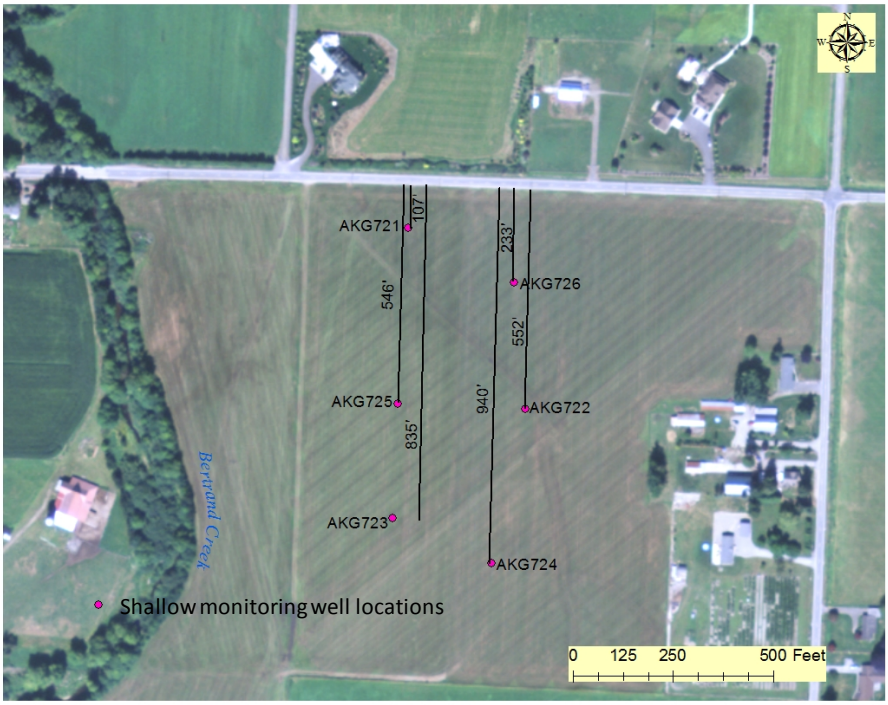

Figure 48. Distances from the edge of the study field to shallow monitoring wells.

Management factors

Rate of nitrogen application - external loading

The amount of nitrogen applied to the ground (from the combination of manure and inorganic fertilizer) has a significant effect on the amount of nitrogen available for leaching to groundwater. In general, the more nitrogen added to a field during the growing season in excess of the crop uptake and removal, the higher the amount of nitrate reaching the water table (Figure 49).

From 2004 through 2007, groundwater nitrate concentrations decreased steadily as a result of decreasing nitrogen load. Chloride, an indicator of nitrate loading, also decreased steadily until late 2008 (Figure 49). Both chloride and nitrate increased again in early 2008 as the nitrogen load increased. Such a rapid groundwater response to changes in nitrogen loading is due to the shallow depth to water and the high seasonal recharge rate.

The relationship between application rate and groundwater nitrate is not exact, because of the many other factors that play a role in nitrogen fate after manure is applied (e.g., availability of manure nitrogen for plant uptake, weather, crop performance, internal nitrate loading from mineralized organic matter). We compared the groundwater nitrate concentration for the typical

time period when nitrate left over at the end of the growing season would have leached to, and been most evident in, shallow groundwater (November to December) to the annual amount of nitrogen applied to the field (manure and inorganic fertilizer) (Figure 48). The correlation was not significant ($r^2 = 0.178$).

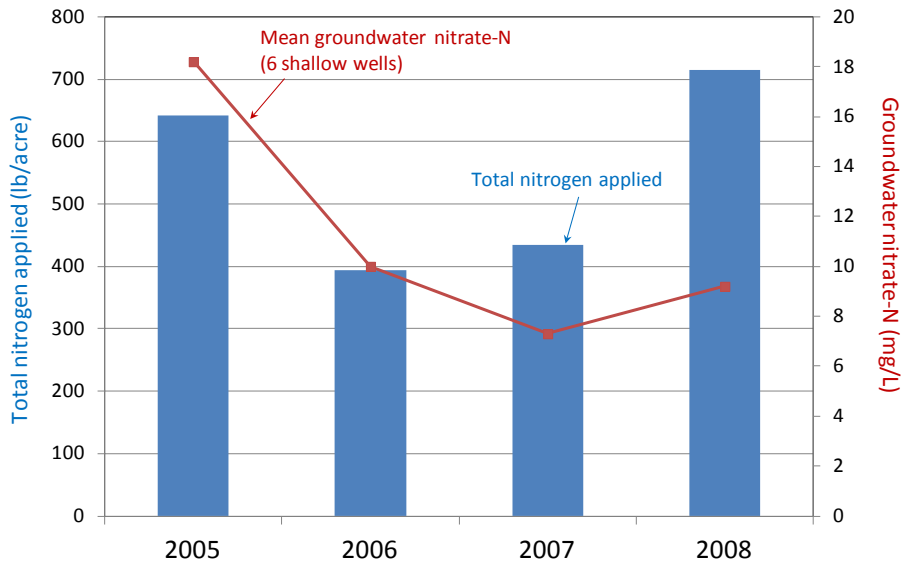

Figure 49. Total nitrogen mass applied (external loading) annually in manure and inorganic fertilizer (bars) and mean early winter (November and December) groundwater nitrate-N concentration in the 6 shallow wells.

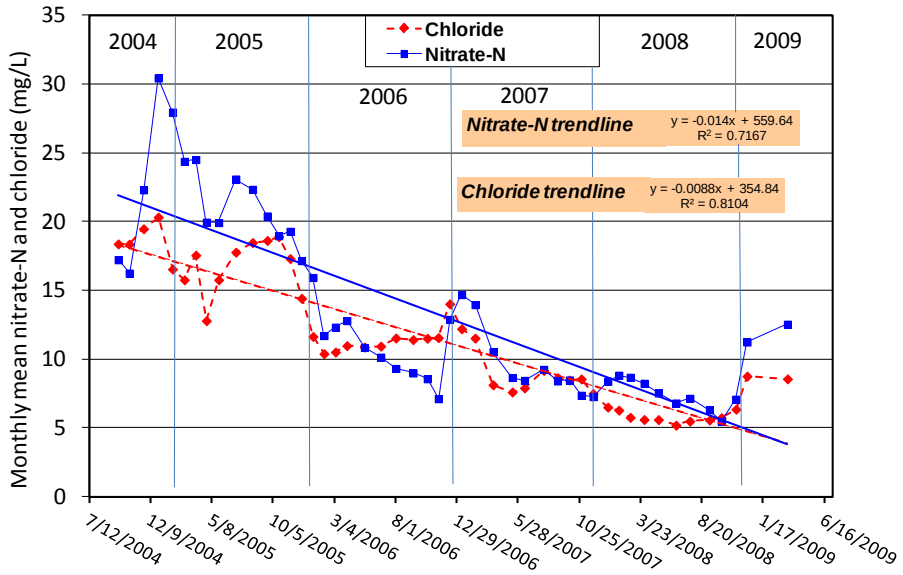

Figure 50. Mean shallow groundwater nitrate-N and chloride concentration trends over time.

The first year that we were able to record nitrogen loading, 2005, was notable for the high rate of external loading. Although unmeasured internal loading contributed additional nitrate to the field that year (due to enhanced mineralization triggered by tillage – discussed below), the high external loading in 2005 was probably the dominant factor affecting subsequent winter groundwater nitrate concentrations. This conclusion is supported by the similarity in groundwater chloride and nitrate trends during the study.

As shown in Figure 50, chloride, an indicator of nitrate loading, decreased steadily from 2004 through 2007. Like nitrate, chloride is a conservative element that does not adsorb to soil particles and is associated with manure application (Rodvang et al., 2004). Manure is the only major source of chloride in the area, and the mass of chloride in the soil that originates from manure would not be affected by tillage (Cogger, 2013). Both chloride and nitrate increased again in early 2008 as the external nitrogen load increased.

Rate of nitrogen mineralization – internal loading

In addition to nitrogen loading to the field from external sources, internal loading via mineralization of organic nitrogen in the soil to nitrate affects the amount of nitrogen available for leaching to groundwater (see *Nitrogen Cycle* section above). It is difficult to accurately quantify the amount and fate of nitrate produced by mineralization.

Tillage effects on mineralization

The highest groundwater and soil nitrate concentrations occurred during the fall and winter of 2004 to 2005. Although this was largely driven by a high rate of external loading, mineralization triggered by tillage of the field earlier in 2004 likely contributed to this condition. Tillage often leads to mineralization of accumulated organic nitrogen, resulting in higher than normal loading of nitrate to the underlying groundwater the following year (Rekha et al., 2011; Oenema et al., 2010; Gupta et al., 2004; Goulding et al., 2000; Whitmore et al., 1992).

Researchers have found conflicting results regarding the influence of tillage on nitrate leaching. Gupta et al. (2004) found higher nitrate leaching after tillage on manured fields than on non-manured fields. This was due to the gradual mineralization of organic matter from manure. Di and Cameron (2002) found the opposite result, higher leaching under no-till fields than under conventionally tilled fields.

Gupta et al. (2004) found higher percolation rates in no-till fields but similar nitrogen flux in manured and non-manured fields. The explanation offered for these differences is that a greater number of worm holes in no-till fields may provide more or better preferential flow paths than in tilled fields. The difference in impact between different cultivation methods may also be due to soil physical and biological factors (Rekha et al., 2011).

Plowing and replanting of the study field in spring 2004, after the field had been unplowed for several years, was at least partly responsible for a significant input of nitrate to the water table in late 2004 to early 2005. The highest monthly mean shallow groundwater nitrate concentration observed during the study, 30 mg/L-N, occurred on December 28, 2004 (Figure 50). This occurred 8 months after tillage, following a rise in the water table to within 1.3 to 4.1 feet of the

ground surface (Figures 29, Plate 3). High fall recharge that caused the water table to rise also transported excess nitrate to the water table.

As mentioned previously, manure is the only major source of chloride at the study site and would not be affected by tillage (Cogger, 2013). Therefore, the significant decrease in both nitrate and chloride concentrations in groundwater from 2004 through most of 2008 indicates that release of nitrate from soil tillage was not the only reason for high nitrate concentrations in late 2004 and early 2005 (Figure 49).

Post growing season mineralization

The cumulative impact of soil nitrogen mineralization on groundwater nitrate leaching is difficult to quantify. Although the rate of mineralization is significantly influenced by temperature, it has been shown to be a significant year-round process (Zebarth, 2013; Zhao et al., 2010; Trindade et al., 2001).

Although a portion of newly mineralized nitrate may be taken up by the grass crop during the winter at the study site, rapid downward water movement probably leaches much of this internal load to the water table, 0 to 7 feet below the ground surface (Zebarth, 2013; Kuipers et al., 2012). Nitrate produced through non-growing season mineralization is generally not accounted for in growing-season mass balance analyses or in post-harvest fall soil nitrate sampling. Nitrate formed in soils through this on-going internal process is potentially available for leaching.

During the winters of 2004-2005 and 2007-2008, nitrate results from several shallow wells indicated a nitrate source other than recently applied manure. Chloride concentrations decreased or remained stable between November and February, while nitrate concentrations increased in 3 wells in 2004-2005 and 4 wells in 2007-2008 (Plate 7).

These nitrate increases in shallow groundwater appear to be from an on-site winter source, because nitrate and chloride concentrations typically track closely to one another absent additional loading. Nitrate and chloride additions from growing-season manure applications had most likely already reached the water table. Therefore, these winter nitrate increases could potentially be from soil nitrate that mineralized after the fall. Another possible source for the winter 2004-2005 groundwater nitrate increase could be lingering effects from nitrate mobilized by tillage, which also would not be linked with chloride.

Crop removal

The mean annual mass of nitrogen removed in the crop was 430 lb/acre (standard deviation = 27) (Figure 46, Plate 6). Consistent levels of nitrogen removal by the crop over 4 years indicate that the large range of nitrogen application rates (393-715 lb/acre) did not affect the crop removal of nitrogen (Figure 47, Plate 6).

Timing of manure applications

Timing of manure application had an effect on nitrate loss to groundwater. Manure applied just prior to major precipitation events at rates that resulted in excess soil nitrate were often followed first by higher soil nitrate values and then by higher shallow groundwater nitrate concentrations.

However, when manure was applied during relatively dry periods and in amounts that the crop could take up, no subsequent increase in soil or shallow groundwater nitrate was observed.

Precipitation at the end of the growing season typically triggers mineralization of soil and manure organic nitrogen sources even without manure application. Examples of manure application timing effects in spring and fall are described below.

Spring applications

Spring application of manure (February through April) can cause spikes in underlying groundwater nitrate concentrations if a large amount of precipitation occurs after the application. The timing and amount of spring manure application over the Sumas-Blaine Aquifer require extreme caution from a groundwater protection standpoint, because recharge is still occurring. Applying manure before the crop can efficiently take up nitrogen in the earliest days of the growing season risks leaching substantial nitrate to groundwater.

In the spring of 2005, although conditions were dry for several days before and after 2 spring manure applications, the total amount of nitrogen applied, 272 lb/acre, combined with nitrate mineralizing from soil organic matter, appears to have exceeded the new grass crop's uptake potential. Groundwater nitrate concentrations increased in all shallow monitoring wells by 4 to 13 mg/L-N during the following 3 months (Plate 8).

Wet weather in March and April 2005 (7 inches of rain in the 2 months following the February 18 application) contributed to downward water and nitrate movement as indicated by the 2.5-foot rise in the water table to within 2.4 feet of the ground surface at well AKG722 (Figure 28, Plate 3). When the water table is close to the surface, most of the nitrate in the soil is probably lost to groundwater (Plate 8).

In the spring of 2007, although the amount of manure nitrogen applied (240 lb/acre) was similar to the amount applied in spring 2005, a smaller and less immediate increase in groundwater nitrate was observed at 3 wells (AKG721, AKG723, and AKG727) (Plate 8). Nitrate concentrations in these wells increased by 2 to 5 mg/L following the 2007 spring manure applications. Precipitation during the month following the initial application, 8 inches, was similar to that in 2005, although evapotranspiration later in the year would have been higher. The smaller increase in groundwater nitrate following the spring 2007 manure applications compared to 2005 may be due to higher evapotranspiration and plant uptake as well as other factors related to the one-month later start date (March 14) in 2007.

Groundwater nitrate increases were less widespread following spring manure applications in 2006 and 2008 (Plate 8). In 2006, less nitrate reached the groundwater due to a combination of a smaller load than in 2005 and 2007 (171 lb/acre nitrogen), application later in the season (April 27), and essentially no recharge the month following application. Lower recharge in spring 2008 than in other years apparently prevented or delayed leaching of nitrate from the relatively heavy spring application of 364 lb/acre nitrogen in 2 episodes (March 10 and April 20).

Wet soil conditions in the spring can delay initial manure application as well as crop harvests throughout the growing season. In 2006, wet weather delayed the first grass cutting until

April 2. The first manure application for the year therefore occurred 6 to 10 weeks later than in previous years. The time available for the repeated process of manure application, crop uptake, and crop removal was thereby shortened by several weeks. This led to a late final manure application in 2006 (October 5) followed by an increase in groundwater nitrate.

Fall applications

In the coastal Pacific Northwest, application of manure in the fall, even on perennial crops, presents a high risk of nitrate leaching (Kowalenko, 1987; Zebarth et al., 1998). In the fall, crop growth slows, precipitation increases, and leaching of nitrate is all but assured.

The consequences of applying manure too late in the fall are demonstrated in Plate 9. For example, a late final fall manure application in October 2006 resulted in an average increase in nitrate concentration of 6 mg/L-N in the shallow winter groundwater (maximum increase of 16 mg/L-N). In 2007, however, when the last application occurred earlier in the season (September), no significant increases in winter groundwater nitrate concentrations were observed. In fact, nitrate decreased by an average of 2.4 mg/L in 5 out of 6 shallow wells.

Most of the variables for nutrient uptake and application were similar in 2006 and 2007. The nitrogen application rate, temperature (growing degree days as defined in Equation 8), and crop removal were similar, although weather and crop removal were slightly better in 2007 (Figures 45 and 46 in Plate 6 and Figure 49). This suggests that the key difference between these 2 years was timing of the final manure application.

2006—Effects on groundwater from late fall manure application

Although the amount of nitrogen applied in the fall of 2006 was the second lowest fall manure application during the study (90 lb/acre), application late in the season on October 5 and pre-existing high soil nitrate levels led to significant effects on shallow groundwater nitrate concentrations.

By November 8, 2006, the soil nitrate concentration increased from 31 mg/kg (96 lb/acre) on October 1 to 60 mg/kg (186 lb/acre), 4 times the recommended level for nutrient balance and at a time well after the typical growing season (Figure 51, Plate 6). A 4.7-inch rain event in early November provided the recharge necessary to transport the nitrate downward. By November 15, 2006, the soil nitrate concentration at 1-foot depth decreased to 15 mg/kg (47 lb/acre), indicating significant leaching of nitrate from the root zone.

As the nitrate mass infiltrated downward in the fall of 2006, the water table rose 6.7 feet (Figure 28, Plate 3). Between October 18 and December 12, 2006, the mean shallow groundwater nitrate concentration increased by over 4 mg/L to 12.9 mg/L-N (Figure 51, Plate 6). Individual well nitrate concentrations increased by up to 16 mg/L-N (Plate 9). Simultaneous increases in chloride concentrations in individual wells indicate manure as the likely source of the nitrate increase (Plate 9).

Prolonged precipitation in the fall can also prevent crop harvest. In 2005, 102 lb/acre of nitrogen in the grass crop were not removed from the field due to wet weather. In such cases, the full

benefit of crop uptake is not realized, because some of the crop nitrogen that would have been removed decays and may become available for leaching to groundwater.

2007—Effects on groundwater from early fall manure application

Early application of manure (September 7) and the lightest fall application during the study, 77 lb/acre, apparently allowed for crop uptake of nutrients before temperatures decreased substantially and before the onset of heavy rain in 2007. As a result, groundwater nitrate concentrations did not spike in the fall of 2007 as they did in 2006 (Plate 9). Nitrate concentrations in 4 out of 6 wells remained below 10 mg/L-N through February 2008.

These examples indicate that timing of manure applications during periods at the margins of the major growing season (fall and late winter) pose a high risk of nitrate leaching due to the combination of unpredictable influencing factors. Precipitation, especially heavy rain, during these times can transport mineralized nitrate (from current and past manure applications) below the root zone before crop uptake can occur.

Conditions during the late winter/early spring are particularly conducive to rapid leaching of available nitrate to the water table. Ammonium from manure applied during this high-recharge time eventually nitrifies to nitrate, while at the same time organic nitrogen remaining in the soil begins to mineralize to nitrate. Nitrate from both sources (winter/early spring manure application and mineralized organic matter) is susceptible to leaching before the grass crop can take up the bulk of the load (Trindade et al., 2001; Zhao et al., 2010; Zebarth, 2013).

Soil moisture

During the growing season, insufficient soil moisture inhibits crop growth. When crop growth is inhibited, less nitrogen is taken up by the crop. This leads to an excess of nitrogen in the soil that can leach to groundwater. In the summers of 2005 and 2006, soil moisture declined to levels that restrict grass growth and bacterial mineralization of ammonium and organic nitrogen (i.e., less than 20%). During this time, the grass crop probably went dormant, resulting in lower nitrogen uptake, lower yield, and more excess nitrogen than if more moisture had been available (VanWieringen and Harrison, 2009) (Figure 17, Plate 2).

In 2007, the dairy producer began irrigating in July, one month earlier than previously, to prevent crop dormancy and improve crop uptake of nitrogen. As a result, soil nitrate concentrations were lower and the crop uptake and removal were higher in the fall of 2007 than in 2006 (Figures 25 and 26, Plate 2).

While additional moisture in the summer of 2007 apparently resulted in higher crop uptake of nitrogen, too much irrigation water during the growing season can move nitrate below the root zone even when the soil is not saturated. This prevents possible crop uptake and contributes to higher groundwater nitrate concentrations. This may have occurred in July and/or August 2008, when the growing season was cooler and wetter than normal.

Denitrification

The dissolved oxygen (DO) condition in soil and groundwater can have a significant effect on groundwater nitrate concentration. When the groundwater DO is below 2 mg/L, denitrification

can remove nitrate and lead to lower nitrate concentrations (Van Es et al., 2006; Rodvang et al., 2004). DO in groundwater varied spatially and over time in the shallow monitoring wells, indicating variability in the rate and timing of denitrification in groundwater.

Denitrification probably caused routine, periodic nitrate loss in groundwater in 4 of the 6 shallow monitoring wells, when the DO was 2 mg/L and lower (AKG722, AKG723, AKG724, and AKG-726 – Plate 8). Denitrification probably also occurred above the water table in these areas before leachate reached the water table, which is typical of wet, fine-textured soils (Coyne, 2008; Paul et al., 1997; Murray et al., 1995). Denitrification above and below the water table therefore likely muted the effect of excess nitrogen infiltrating to groundwater, especially from the wells with low DO.

Nitrate: chloride ratio as an indicator of denitrification

As described earlier, chloride is largely non-reactive in the subsurface. Changes in chloride concentrations in groundwater are therefore assumed to be due to either dilution or a change in chloride loading. These characteristics make chloride useful for evaluating nitrate changes in groundwater (McCallum et al., 2008; Rodvang et al., 2004).

An additional tool for evaluating whether denitrification is a major factor in controlling groundwater nitrate concentrations is to compare ratios of nitrate-N to chloride (NO_3-N:Cl) in groundwater (McCallum et al., 2008). Variations in the ratio of nitrate-N to chloride can indicate loss of nitrate due to denitrification where DO is sufficiently depleted.

Nitrate and chloride consistently followed similar patterns in the 2 high-DO wells (AKG721 and AKG725; Figure 52, Plate 6). Initially both wells had a higher concentration of nitrate than chloride. This was probably due to increased mineralization of soil organic matter related to tilling the field. After the first year, however, nitrate and chloride concentrations corresponded more closely.

During months with little or no recharge, and when the water table is too far below the root zone for plant uptake (June through September), relative differences in the proportion of nitrate and chloride concentrations (NO_3-N:Cl) are probably due to denitrification rather than to crop uptake or management activities.

Nitrate and chloride concentrations generally tracked closely when DO was above the 2 mg/L threshold for denitrification in the 4 wells with at least occasionally low DO (Figure 53, Plate 6). However, when DO fell below 2 mg/L, nitrate and chloride concentrations diverged, indicating loss of nitrate to denitrification. For example, in the fall of 2006, when DO in AKG722, AKG723, AKG724, and AKG727 was below 2 mg/L (Plate 9), chloride concentrations remained at the same level or increased. During the same time period, nitrate concentrations dropped by 4-10 mg/L-N (Figure 53, Plate 6).

The mean NO_3-N:Cl ratios for the 2 DO conditions were:

- High-DO wells (always greater than 2 mg/L): 1.39 (SD[9]=0.26, n=96)

[9] Standard deviation

- Low-DO wells (less than 2 mg/L at times): 1.05 (SD=0.44, n=188)

The NO_3-N:Cl ratio was 32% higher in the high-DO wells than in the low-DO wells throughout the study and was relatively stable. The higher variation, as indicated by the higher standard deviation, in the seasonally low-DO wells is consistent with the fluctuation in DO concentrations above and below the denitrification threshold.

The nitrate losses indicated by the NO_3-N:Cl ratio suggest an average 28% reduction in nitrate in the low-DO wells compared to the high-DO wells. For example, if 2 groundwater samples from the site each contained 10 mg/L chloride, the sample from a high-DO well would have a nitrate concentration of 13.9 mg/L-N on average. The sample from a low-DO well with 10 mg/L would have a nitrate concentration of 10.1 mg/L-N on average.

McCallum et al. (2008) found a similar distinction between NO_3-N:Cl ratios in groundwater beneath manured fields where denitrification was occurring and manured fields where denitrification was not occurring.

Summary of major influences on groundwater nitrate conditions at the study site

The management and environmental factors discussed above interact and influence both the amount of nitrate that leaches to the water table and the resulting groundwater nitrate concentrations in a complex manner. The data collected during this study suggest that groundwater nitrate conditions at the study site were particularly sensitive to:

- Manure applications that significantly exceeded crop demand.
- Manure applications that occurred outside of the typical growing season, particularly during periods of high recharge.
- Initial tillage event.
- Denitrification capacity of the aquifer.

Annual nitrogen balance and residual estimation – correlation to groundwater quality

A primary goal of Dairy Nutrient Management Plans (DNMPs) is to closely balance nutrient application and plant uptake to minimize residual or excess nitrogen remaining at the end of the growing season. Residual nitrogen in nitrate form can be transported to the water table, where it can impact groundwater quality (Harter and Menke, 2005; Zebarth et al., 1998).

Two field-based methods for evaluating the end-of-growing-season nitrogen residual are:

1. Indirect estimation by mass balance analysis.
2. Direct estimation by end-of-season soil sampling.

Below we discuss annual nitrogen residuals predicted by these 2 approaches and compare the estimates to observed groundwater conditions beneath the study field.

1. Estimate of annual nitrogen residual by mass balance analysis

The various nitrogen inputs and outputs measured during the study can be incorporated into a mass balance analysis to estimate the residual amount of nitrogen mass present at the end of each growing season using Equation 12. The method assumes that measurements of nitrogen inputs and outputs are complete and that there are no unidentified gains, losses or storage in the system. In cases where an equation variable was not directly measured in the field, an estimate for the value was developed using standard industry assumptions. This method also assumes that all excess nitrogen is converted to the nitrate form and leaches to groundwater at the end of the growing season.

$$N_{Excess} = N_{Input} - N_{Output} \qquad (Eq.\ 12)$$

where:
N_{Excess} = Excess nitrogen mass left in the soil column at the end of the growing season (lbs/acre)
N_{Input} = Total nitrogen mass input during the growing season (lbs/acre)
N_{Output} = Total nitrogen mass output during the growing season (lbs/acre)

$$N_{Input} = PAN + F + A + I \qquad (Eq.\ 13)$$

where:
PAN = Plant-available nitrogen from manure and soil organic matter (lbs/acre—calculated based on measured manure N)
F = Nitrogen applied as inorganic fertilizer (lbs/acre, estimated)
A = Nitrogen from atmospheric deposition (lbs/acre, estimated)
I = Nitrogen applied in irrigation water (lbs/acre, measured and estimated)

and

$$N_{Output} = Y + V + D + L \qquad (Eq.\ 14)$$

where:
Y = Nitrogen removed in crop (lbs/acre, measured)
V = Nitrogen lost to volatilization during application (lbs/acre, estimated)
D = Nitrogen lost to denitrification (lbs/acre) (term ignored for this analysis)
L = Nitrogen lost to leaching during the growing season (lb/acre) (term ignored for this analysis)

Nitrogen losses due to denitrification in the unsaturated zone (above the water table) were considered negligible during the growing season (Sullivan, 2008); therefore, the variable D in Equation 14 was ignored. Leaching losses were also assumed to be negligible for the analysis, because recharge is very low during the growing season. Therefore, variable L in Equation 14 was also ignored.

Three methods, described below, were used to estimate plant-available nitrogen (PAN), the main component of N_{Input} in Equation 12.

Method 1-PAN

Method 1 for estimating PAN is based on Sullivan's (2008) method, as shown in Equation 15.

$$PAN = 0.85M_{NH4} + 0.4(M_{OrgN}) + 0.15(OrgN_{Yr\,1}) + 0.07(OrgN_{Yr\,2}) + 0.03(OrgN_{Yr\,3}) + 0.02(OrgN_{Yr\,4-9\,Sum})$$

(Eq. 15)

where:
M_{NH4} = ammonium-N content of the manure applied during the current year (lbs/acre)
M_{OrgN} = organic N content of the manure applied during the current year (lbs/acre)
$OrgN_{Yr\,1}$ = organic N content of the manure applied the year before the current year (lbs/acre)
$OrgN_{Yr\,2}$ = organic N content of the manure applied 2 years before the current year (lbs/acre)
$OrgN_{Yr\,3}$ = organic N content of the manure applied 3 years before the current year (lbs/acre)
$OrgN_{Yr\,4\text{-}9\,Sum}$ = organic N content of the total amount of manure applied during the period 4 to 9 years before the current year (lbs/acre)

Equation 15 assumes that during the current year, 85% of the ammonium-nitrogen and 40% of the organic nitrogen in the manure is available to plants when applied using surface banding. An increasingly smaller portion of the organic nitrogen is available from previous applications up to year 4 before the current year, when the portion remains constant at 2% through year 9.

The mean annual organic nitrogen in manure during the study, 275 lb/acre/year, was used to estimate the nitrogen mineralized from organic matter for subsequent years (Appendix J, Table J.2).

Variable (*V*) in Equation 14 is ignored in the mass balance analysis using Method 1, because loss of ammonia to volatilization is accounted for in the method. The Method 1 PAN estimates developed for each study year are shown in Table 9.

Table 9. PAN estimates (lb/acre) using 3 methods.

PAN Method	2005	2006	2007	2008	Average
1	517	310	321	530	420
2	565	359	369	578	468
3	644	363	386	715	527

Equation 15 assumes that manure applied during the current year is in the form of a thin slurry (1 to 5% dry matter) and is applied using surface banding. Liquid manure applied during the study was 1 to 5% dry matter on 16 out of 17 dates and 7% on one date.

Method 2-PAN

Method 2 for estimating PAN uses a modification of the Sullivan (2008) procedure to account for the possible effects of relatively high organic matter measured in the study site soils (average 7.5%) (Zebarth, 2013; Sullivan, 2013).

Method 2 estimates the contribution of PAN from previous years' applications using either Equation 16 or 17:

When the organic matter in the soil is less than 5% (NRCS, 2006):

$$PAN = 0.85M_{NH4} + 0.4M_{OrgN} + 20(OM) \qquad Eq.16$$

or, when the organic matter in the soil is equal to or greater than 5% (NRCS, 2006):

$$PAN = 0.85M_{NH4} + 0.4M_{OrgN} + 150 \qquad Eq.17$$

where:
OM = average organic matter of the soil (%)

Method 2 also takes into account loss of ammonia to volatilization; therefore, variable (*V*) in Equation 14 is ignored for the mass balance analysis for this scenario. PAN estimates for Method 2 are shown in Table 9.

Method 3-PAN

Method 3 for estimating PAN uses the total nitrogen applied in manure for the year (Equation 18). Although not all of the total nitrogen applied is actually available that year, this method assumes that the mass of nitrogen that is not plant-available is balanced out with the amount that is mineralized from soil organic matter. Method 3 estimates for PAN are shown in Table 9. In contrast to PAN estimation Methods 1 and 2, the Method 3 PAN values do not account for ammonia volatilization; therefore, a separate value for variable (*V*) in Equation 14 is included in the mass balance analysis. (See *Other elements of the nitrogen balance* below.)

$$PAN = Total\ N_{Manure} \qquad Eq.18$$

where:
$Total\ N_{Manure}$ = Total mass of nitrogen applied in manure for the year

Comparison of 3 methods for estimating nitrogen inputs

The 4-year mean for the 3 methods for estimating nitrogen inputs are shown in Figure 54. Methods 1 and 2 produced roughly the same mean value, 418 to 420 lb/acre, while Method 3 was 25% higher. Standard deviations for the 3 input methods were 24 to 34% of the means (or 98 to 179 lb/acre).

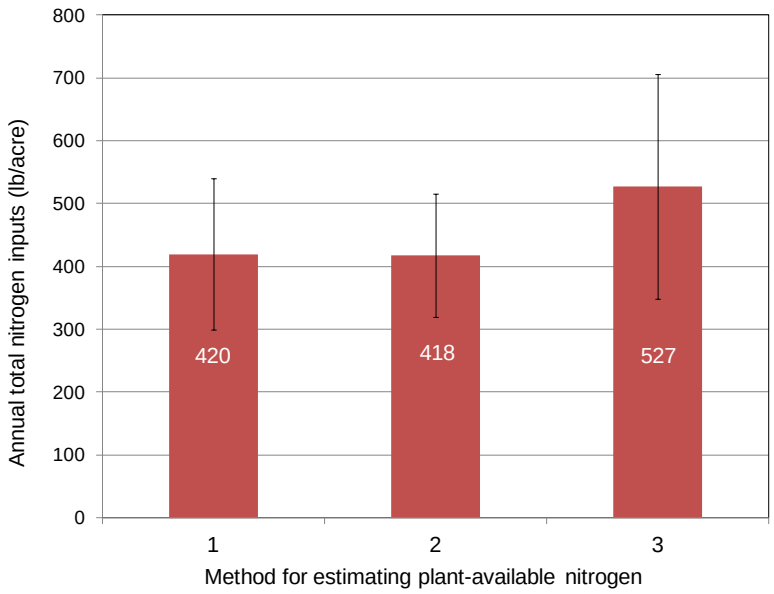

Figure 54. Mean annual nitrogen inputs estimated using PAN Methods 1 through 3.
Error bars indicate +/- one standard error. Numbers on the bars are the mean values.

There is considerable uncertainty associated with each method for calculating the PAN component of N_{Input}. The most difficult PAN value to quantify is the amount of nitrogen mineralized from the complex mixture of organic compounds in manure and soil (University of California, 2007). While it is well documented that organic forms of nitrogen mineralize in a half-life decay pattern, with slower breakdown of more complex compounds over time, many factors affect the rate of decay. The estimates for nitrogen available from soil organic matter used in this analysis probably have a large margin of error.

The simplified manner in which Method 1 was applied (i.e., using a single value to represent the organic nitrogen content across multiple years) does not capture the variability in manure composition and loading within each year or between years. The value used in Method 1 for estimating PAN from soil organic matter, the mean annual organic nitrogen mass (275 lb/acre), may not represent the actual variation in organic nitrogen content in the manure applied in past years. The largest component of PAN in Method 1 (67 to 81%), however, comes from the current year, making past contributions less influential.

Method 2 provides an even less reliable estimate of the soil organic nitrogen contribution to the PAN pool than Method 1. Significant uncertainty exists regarding whether soil organic matter is a reliable indicator of PAN, because the organic matter in soils with the highest organic matter content is older and tends to decompose more slowly than organic matter in soils with lower percentages (Sullivan, 2013). Soils with lower organic matter content are typically of more recent origin, and this organic matter tends to mineralize more readily. NRCS (2006) presents a wide range for nitrogen mineralization in soils with 5% or greater organic carbon (50 to 200 lb/acre).

The soil organic nitrogen contribution to PAN in Method 3 is based on agricultural experience, but we were not able to confirm how well results for this method compare with field conditions.

Average PAN input using this method was 107 and 59 lb/acre higher than Methods 1 and 2 respectively (Table 9).

Other elements of the nitrogen balance

In addition to PAN, other elements measured during the study include nitrogen inputs from inorganic fertilizer and irrigation water (Equation 13) and nitrogen outputs from crop removal (Equation 14). Components that were not measured during the study [atmospheric deposition and volatilization (for Method 3 only)] were estimated based on the following assumptions:

- Annual atmospheric input of nitrogen was assumed to be similar to that reported at nearby Abbotsford, British Columbia, 8 lb/acre/year (Kuipers et al., 2012).
- For the Method-3 PAN scenario, 15 % of the annual PAN was assumed to be lost to ammonia volatilization. This assumes that subsurface deposition is equivalent to surface banding (partial incorporation) (Sullivan, 2008).

The estimated values for atmospheric inputs and ammonia volatilization for each year are presented in Table 10. The contribution of nitrate to the soil and groundwater from tilling the field in April 2004, due to enhanced mineralization of soil organic matter, was probably substantial (Oenema et al., 2010; Goulding et al., 2000; Whitmore et al., 1992).

Mass balance results – annual estimates of residual nitrate

Table 10 summarizes the annual N_{Input} and N_{Output} estimates for 2005 to 2008 using the 3 PAN estimation methods described above. The table also presents the N_{Excess} estimate for each year calculated using Equation 12. A mass balance analysis was not conducted for 2004, because detailed information on manure nitrogen inputs was not available.

The annual N_{Excess} estimates ranged from -79 to 146 lb/acre. The 4-year average mass balance residuals were 19 lb/acre for Method 1, 67 lb/acre for Method 2, and 79 lb/acre for Method 3.

The estimated annual nitrogen inputs and outputs are shown graphically in Figures 55 (inputs only) and 56, using the Method 1-PAN values. Because the Method 1-PAN estimation approach is based on the most up-to-date regional understanding of nitrogen dynamics in manured fields, subsequent analyses in this report will use only results of this method. Our qualitative judgment of the level of confidence in each input category is shown in the Figure 54 legend.

The mass balance analysis indicates:

- Approximately one-half of the nitrogen input comes from ammonium directly applied to the crop.
- Internal loading by mineralization of organic nitrogen contributes nearly as much as direct application.
- Other sources such as irrigation and atmospheric inputs represent only a small fraction of the overall nitrogen load.
- Crop removal is the dominant process for nitrogen output during the growing season.
- The Method 1 mass balance analysis indicates that substantial nitrogen deficits occurred during the 2006 and 2007 study years.

Table 10. Mass balance calculation of annual nitrogen (N) excess at the end of the growing season (lb/acre).

Method 1 - PAN	2005[1]	2006	2007	2008	Average
INPUTS (N_{Input})					
PAN from Sullivan (2008) (M)	517	310	321	530	
Inorganic fertilizer (F)	0	31	48	0	
Irrigation water (I)	0.8	1.8	1.7	1.4	
Atmospheric input (A)[2]	8	8	8	8	
N_{Input} totals	526	351	379	539	
OUTPUTS (N_{Output})					
Crop N removed (Y)	439	430	457	393	
N_{Output} totals	439	430	457	393	
N_{Excess} = (N_{Input} - N_{Output})	**87**	**-79**	**-78**	**146**	**19**
Method 2 - PAN	**2005[1]**	**2006**	**2007**	**2008**	
INPUTS (N_{Input})					
PAN from Sullivan (2008) for current year + 150 lb/acre SOM[3] (M)	565	359	369	578	
Inorganic fertilizer (F)	0	31	48	0	
Irrigation water (I)	0.8	1.8	1.7	1.4	
Atmospheric input (A)[2]	8	8	8	8	
N_{Input} totals	574	400	427	587	
OUTPUTS (N_{Output})					
Crop N removed (Y)	439	430	457	393	
N_{Output} totals	439	430	457	393	
N_{Excess} = (N_{Input} - N_{Output})	**135**	**-30**	**-30**	**194**	**67**
Method 3 - PAN	**2005[1]**	**2006**	**2007**	**2008**	
INPUTS (N_{Input})					
Manure total nitrogen applied (M)	644	363	386	715	
Inorganic fertilizer (F)	0	31	48	0	
Irrigation water (I)	0.8	1.8	1.7	1.4	
Atmospheric input (A)[2]	8	8	8	8	
N_{Input} totals	653	404	444	724	
OUTPUTS (N_{Output})					
Crop N removed (Y)	439	430	457	393	
Ammonia volatilized (15% of Ammonium-N applied) (V)	62	31	33	64	
N_{Output} totals	501	461	490	457	
N_{Excess} = (N_{Input} - N_{Output})	**152**	**-57**	**-46**	**267**	**79**

[1] Last grass cutting, 102 lb/acre, was not removed from the field but is included in crop nitrogen removal.
[2] From Kuipers (2012). [3] Soil organic matter.

Despite substantial estimated nitrogen deficits in 2006 and 2007, annual crop nitrogen removed from the field was roughly the same each year (Figure 46, Plate 6). The highest yield, 457 lb/acre N, was obtained in 2007, when the Method 1-PAN indicated the highest deficit, -79 lb/acre. The lowest yield observed, 393 lb/acre N in 2008, followed the highest annual Method 1-PAN mass balance excess (146 lb/acre). This indicates that plant nitrogen uptake was probably not limited by nitrogen application and suggests that more nitrogen was available for both plant uptake and leaching than indicated by the mass balance analysis.

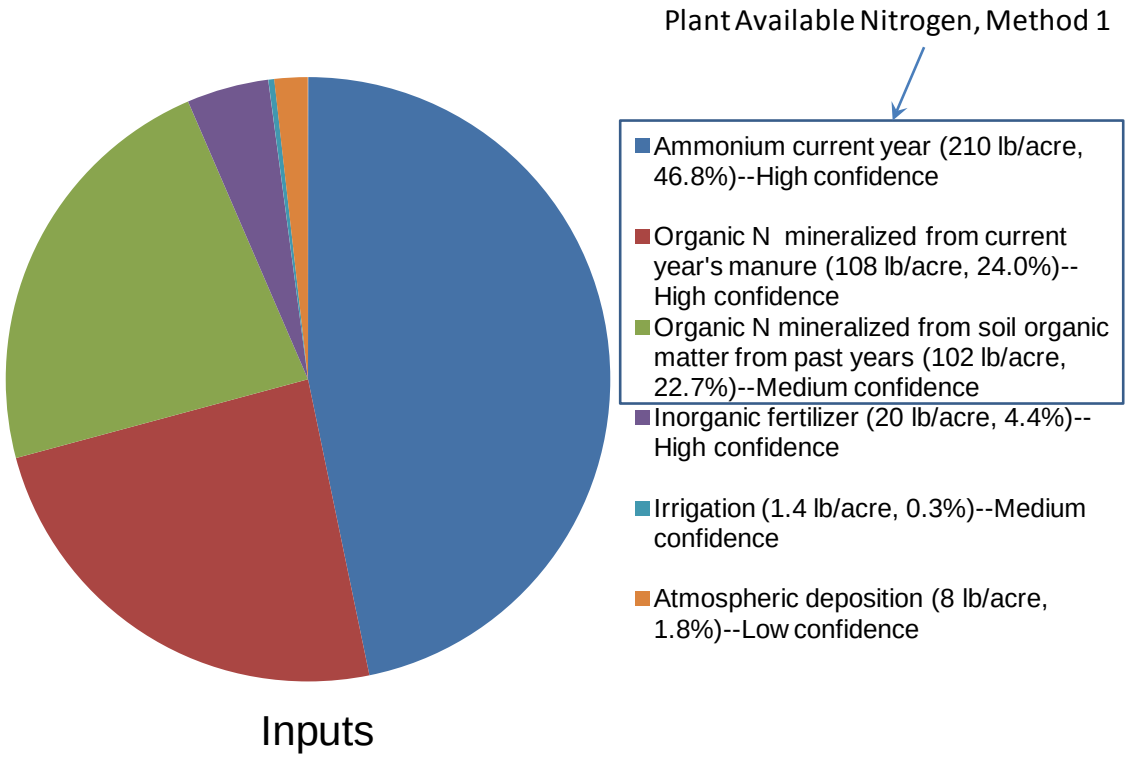

Figure 55. Average annual nitrogen inputs to the soil column during the study in lb/acre using Method 1 and our relative degree of confidence in the numbers.

Data are shown in Appendix S.

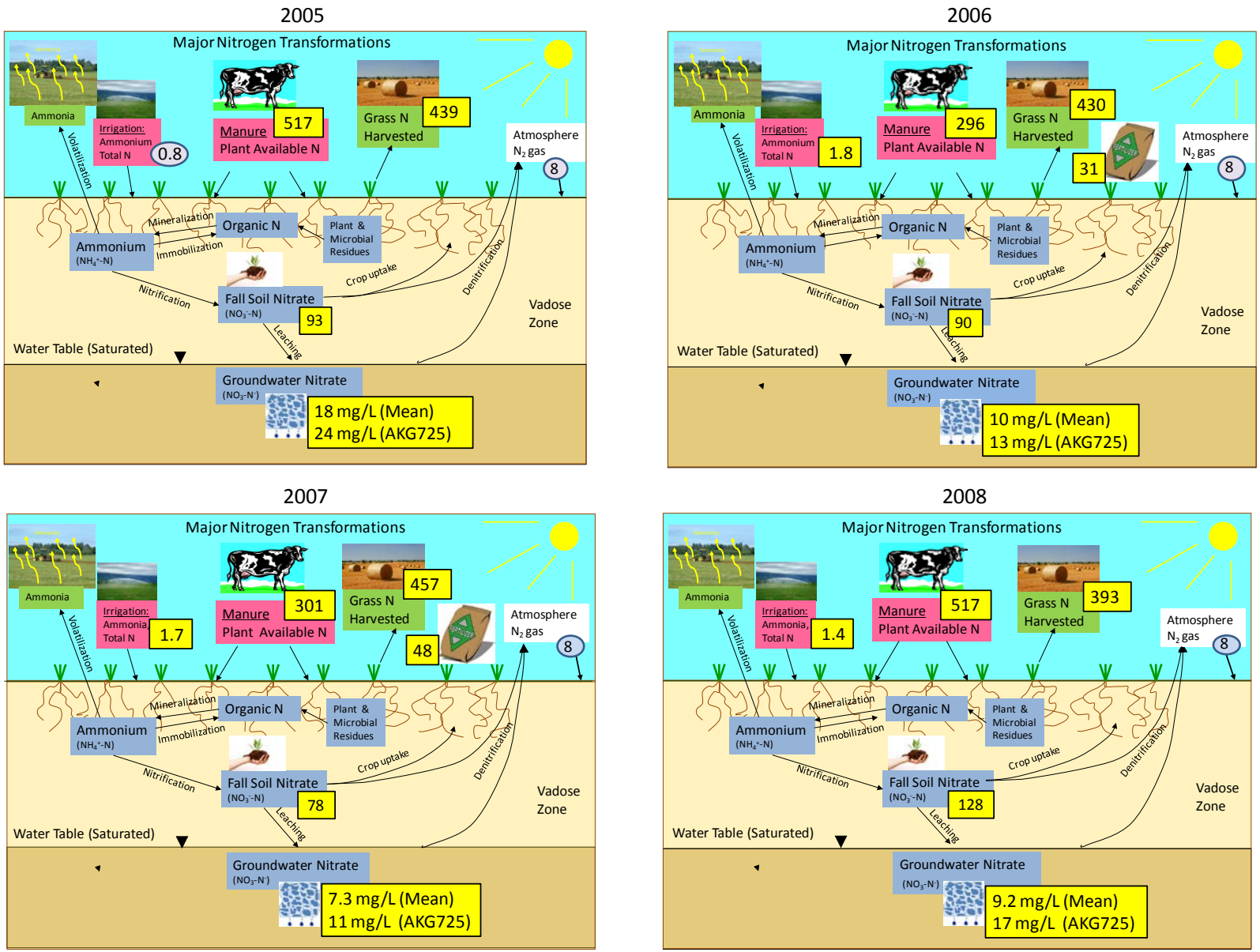

Figure 56. Annual nitrogen inputs, outputs, and soil nitrate residual.

See next page for explanation.

Explanation for Figure 56: Inputs, outputs, and soil nitrate are in lb/acre. Values in yellow boxes were measured. Values in blue ovals are estimates. Groundwater nitrate-N concentrations (Mean) represent mean winter (November to December) values for the 6 shallow monitoring wells are in mg/L; (AKG725) represents mean winter values for well AKG725. Plant Available N (shown in the Manure box) is from Method 1 in Table 9. Atmospheric N input is from Kuipers et al. (2012). Fall soil nitrate residual is the maximum of weekly values for September 1 through October 31. Denitrification outputs from soil are assumed to be negligible during the growing season.

Color code: green=outputs, pink=inputs, white=inputs and outputs, blue= resulting effects on soil and groundwater.

Sources of uncertainty in the N_{Excess} results

Most of the nitrogen inputs during the growing season were measured with high precision and accuracy and are based on direct field monitoring results (Figure 55). Outputs in the form of crop removal were also measured precisely and accurately. However, the methods for estimating a number of the other components of the mass balance have important limitations and therefore introduce a degree of uncertainty in the residual estimates. The major sources of uncertainty in the mass balance analysis include:

- The 2005 N_{Excess} estimates are biased low, because the last cutting could not be removed from the field. A portion of the nitrogen in the grass decomposed and became available for leaching but is not included in the Table 10 calculations.
- The amount of nitrogen mineralized from soil organic matter is difficult to quantify and can be substantial (Sullivan, 2013).
- The organic matter concentration used in the PAN estimates for past years was based on the mean organic nitrogen concentration in manure during the study and may be biased either high or low.
- Atmospheric deposition at the site could be higher or lower than the rate used, which would bias the N_{Excess} results either high or low.
- The amount of nitrate lost to leaching during the growing season following irrigation or heavy precipitation events may have been significant.

2. Estimate of annual nitrate residual using soil sampling results

Although the results show that the timing of sample collection can have a significant effect on the observed result, direct measurement of soil nitrate concentrations at the end of the growing season is a standard industry method for estimating residual nitrogen.

Equation 19 was used to convert fall soil nitrate sample concentration results from mg/kg dry weight to residual nitrate mass values in units of lb/acre. To make this conversion, Equation 19 requires a value to be set for the bulk density of the soil (in g/cm^3). The bulk density value for Hale silt loam soil (0- to 10-inch depth) at the site is 1.13 g/cm^3, assuming the average organic matter content at the site is 7.5% (average of annual values at the site). Therefore, the conversion factor from mg/kg nitrate dry weight to lb/acre ($\rho_b \; x \; 2.791$) is 3.1. This is somewhat lower than the conversion factor suggested for western Washington (3.5) by Sullivan and Cogger (2003).

$$RN_{Soil} = C_{Soil\ N}\ x\ \rho_b\ x\ 2.79 \qquad (Eq.\ 19)$$

where:
RN_{Soil} = Residual soil nitrate mass at 1-foot depth (lb/acre)
$C_{Soil\ N}$ = Soil nitrate concentration, dry weight (mg/kg DW)
ρ_b = Soil dry bulk density (g/cm^3)
2.791 = Unit conversion constant (to convert mg NO_3-N/kg (dry weight) to lbs NO_3-N/acre-ft)

Table 11 presents both the average and maximum soil nitrate concentration measured at the site each year from September 1 through October 31 (2004-2008). The corresponding residual mass values estimated using Equation 19 are also presented. These residual mass values represent the amount of nitrate that end-of-season soil sampling in the top 1 foot indicated was potentially available to leach to groundwater. The residual N_{Excess} mass estimates determined by the Method 1 mass balance evaluation are also presented in Table 11 for comparison.

Table 11. Average and maximum end-of-season soil nitrate concentrations and estimated residual mass values.

Study Year	Sept-Oct average soil nitrate concentration ($C_{Soil\ N\ AVG}$) (mg/kg)	Nitrate mass residual predicted by average fall soil concentration ($RN_{Soil\ AVG}$) (lbs/acre)[1]	Sept-Oct maximum soil nitrate concentration ($C_{Soil\ N\ MAX}$) (mg/kg)	Nitrate mass residual predicted by maximum fall soil concentration ($RN_{Soil\ MAX}$) (lbs/acre)[1]	Method 1 mass balance residual nitrogen mass estimate (N_{Excess}) (lbs/acre)[2]
2004	28.0	87	43.0	133	N/A
2005	19.9	62	30.0	93	87
2006	20.9	65	29.0	90	-79
2007	16.4	51	25.3	78	-78
2008	30.7	95	41.2	128	146
Average	23	72	34	104	19

[1] Estimates calculated using Equation 19. Soil dry bulk density assumed = 1.13 g/cm^3.
[2] Values from Table 10.

The 4-year average nitrate residual mass values of 72 and 104 lbs/acre estimated from soil samples are approximately 4 to 5 times greater than the 4-year average estimated by the mass balance approach (19 lbs/acre). It is important to note that the root mass of a growing crop also has demand for nitrogen, although we do not have good estimates for this number. This root mass has a growth and death cycle which leads to release of nitrogen in soil.

The nitrate residual mass values in Table 11 are predicted using soil sample results only from the September 1 to October 31 time frame. In some cases, soil nitrate concentrations measured after October suggest significantly higher nitrate residuals. For example, in 2006, the predicted September to October $RN_{Soil\ MAX}$ value was ~90 lbs/acre. However, if the maximum soil measurement collected in November of that year (60 mg/kg) had been used for the calculation, the resulting $RN_{Soil\ MAX}$ estimate would increase to ~186 lbs/acre. This value is more than twice that

calculated with the September to October data and would be nearly 3 times as great as the $RN_{Soil\ AVG}$ value.

The annual nitrate residual mass values predicted by soil sampling are not consistent with the values predicted by the mass balance approach. In some cases, predictions were very close (the 2005 maximum soil nitrate residual vs. the 2005 mass balance predicted residual) as shown in Table 11. In other cases, the estimates were very different (e.g., 2006 and 2007).

Correlation of annual nitrogen residual estimates with groundwater quality conditions

Mass balance nitrogen residual estimates vs. groundwater nitrate concentrations

In order to observe how well the mass balance analysis approach predicted the near-term effect of end-of-season nitrate leaching to groundwater, we compared the annual N_{Excess} to the average late fall/early winter (November 1-December 31) nitrate concentration in the shallow monitoring wells. Despite intensive sampling of nitrogen inputs and outputs more extensive than normally conducted by dairy producers, the end-of-season mass balance nitrogen residual estimates did not correlate with groundwater nitrate concentrations during the higher recharge months immediately following the growing season ($r^2 = 0.19$; Table 12 columns B, C and D; Figure 57)

Table 12. Correlation of nitrogen mass residual estimates to mean November to December groundwater nitrate concentrations.

A	B	C	D	E	F	G	H
Year	**Nov-Dec GW nitrate-N conc. mean (mg/L)[1]**	**Residual nitrogen mass predicted by Method 1 mass balance (N_{Excess}) (lbs/acre)**	**Mass balance N_{Excess} vs. Nov-Dec GW nitrate-N conc. (r^2)**	**Residual nitrogen mass predicted by average soil nitrate conc. ($RN_{Soil\ AVG}$) (lbs/acre)**	**$RN_{Soil\ AVG}$ vs. Nov-Dec GW nitrate-N conc. mean (r^2)**	**Residual nitrate mass predicted by maximum soil nitrate conc. ($RN_{Soil\ MAX}$) (lbs/acre)**	**$RN_{Soil\ MAX}$ vs. Nov-Dec GW nitrate-N conc. mean (r^2)**
2004	26.5	NA		87		133	
2005	18.2	87		62		93	
2006	10.0	-79	0.19	65	0.12	90	0.30
2007	7.3	-78		51		78	
2008	9.2	146		95		128	

[1]Values represent the mean concentration of the 6 shallow wells (AKG721, AKG722, AKG723, AKG724, AKG725, and AKG727) between November 1 and December 31.
GW = groundwater. conc. = concentration.

The difficulty in accurately estimating all components of the mass balance (Equation 12) makes this method unreliable for predicting the effects on groundwater. The uncertainty in the "internal loading" contribution of nitrogen that occurs due to mineralization of soil organic matter is likely a primary explanation for the lack of correlation (Sullivan, 2013; Zebarth, 2013). Storage of nitrogen in the root system of the crop could be another influential factor.

A mean value for multiple annual mass balance residuals (i.e., 3 years) may produce a result closer to the actual amount of nitrate leached. Although we took into account nitrate mineralization from previous years in the mass balance calculation, actual nitrate mineralization from previous manure applications may vary from the method predictions. A longer period of record with more stable inputs and outputs would be needed to test whether a 3-year average mass balance residual coincides better with shallow groundwater nitrate concentrations.

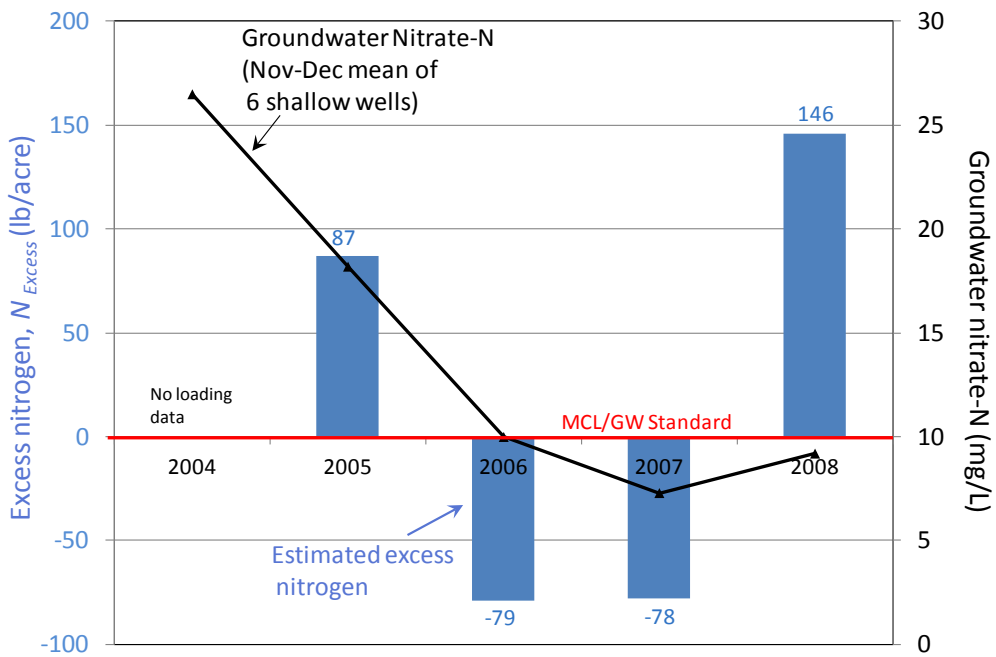

Figure 57. Comparison of annual N_{Excess} (bars) with mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

MCL = maximum contaminant level.

Fall soil nitrate residual estimates vs. groundwater nitrate concentrations

Data were not available for nitrogen application in 2004; however, high soil nitrate concentrations in the fall (up to 160 lb/acre in August) indicated a substantial residual of nitrate was available for leaching during the fall and winter of 2004-2005 (Figure 26, Plate 2). Accordingly, the highest groundwater nitrate concentration observed, 45 mg/L-N, occurred on December 28, 2004.

While soil nitrate concentrations indicated substantial nitrate was available for leaching in the fall each year, neither the $RN_{Soil\ AVG}$ nor the $RN_{Soil\ MAX}$ values were significantly correlated with the mean November-December groundwater nitrate concentrations (Table 12, columns B, E, F, G, and H; Figure 58). This is likely because mineralization and leaching are continual processes. The concentration of nitrate in the soil can only indicate the amount left over at that point in time, with no indication of the amount of nitrate that has already leached or the amount that will become available. This suggests that fall soil nitrate monitoring, even when conducted at a high frequency, is not a reliable predictor of groundwater responses to nutrient management activities.

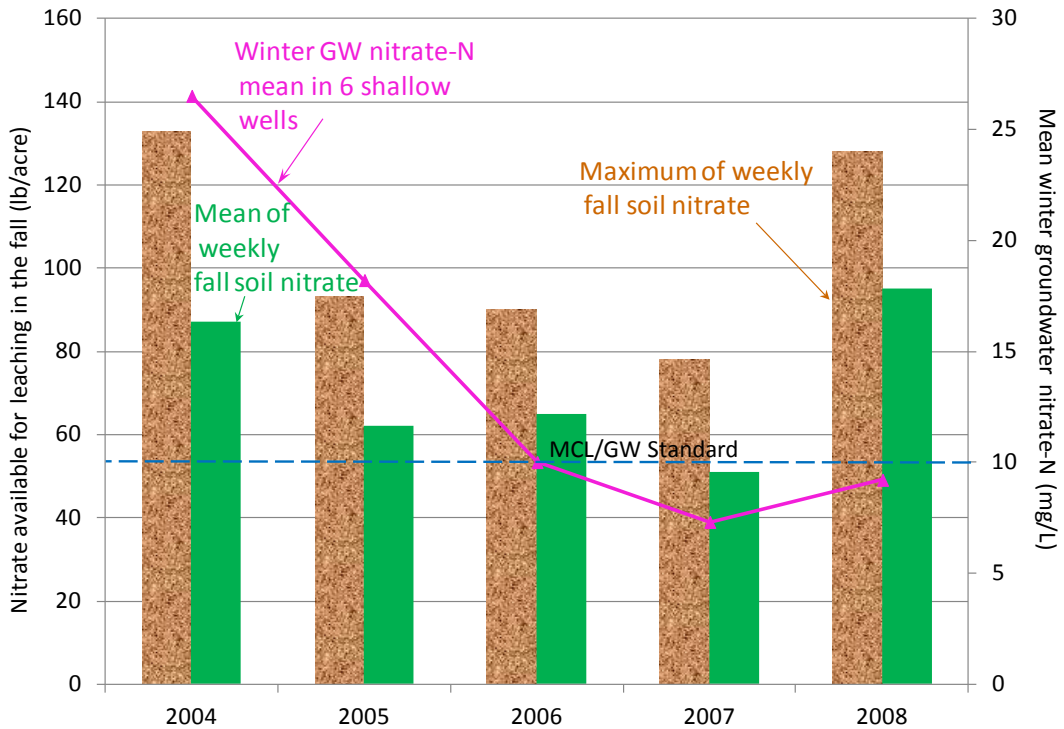

Figure 58. Comparison of annual fall soil nitrate mean concentrations (green, non-textured bars) and maximum concentrations (brown, textured bars) with mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

Groundwater nitrate GWNO3-BACKCAST modeling

Because neither the mass balance nor the fall soil nitrate field methods provided a reliable prediction of groundwater responses to manure management, we used a simplified mixing-box model (GWNO3-BACKCAST; Pitz, 2014, in preparation) as an alternative method for estimating the annual residual nitrogen at the study field.

The BACKCAST model back-calculates the mass load of soluble nitrate (model variable name: $NO3_{TotExcess}$) required to produce a known groundwater nitrate condition (model variable name: $C_{GW\ Outflow\ NO3}$). The $NO3_{TotExcess}$ values predicted by the model are directly comparable to the Method 1 mass balance N_{Excess} residual values and also to the $RN_{Soil\ AVG}$ and $RN_{Soil\ MAX}$ residual values estimated by soil sampling.

Although the BACKCAST model is based on a simplified conceptual model of nitrate loading to an aquifer, one important advantage of the model is that it can account for *saturated_zone* processes such as mixing and denitrification. These processes are not accounted for by the mass balance or soil nitrate sampling approaches. The model can be a useful tool for approximating components of the nitrogen cycle that are otherwise difficult to measure, such as mineralization.

Appendix T presents a summary of the model equations, assumptions, and input data used for each study year. Outputs and sensitivity analysis for each input parameter are also described in Appendix T.

GWNO3-BACKCAST estimates for late fall/early winter nitrate mass loading (Period A)

The BACKCAST model was used to quantitatively predict the late fall/early winter nitrate mass load necessary to generate the average shallow groundwater nitrate-N concentrations observed between November 1 and December 31 each study year (compilation of shallow monitoring well data only). A saturated zone denitrification rate of 15% was assumed for the modeling analysis on the basis of field observations (see *Nitrate: chloride ratio as an indicator of denitrification*). Table 13 summarizes the annual $NO3_{TotExcess}$ predictions for this scenario. Figure 59 graphically compares the annual Period A $NO3_{TotExcess}$ predictions to the annual residual mass estimates produced by the mass balance and maximum fall soil nitrate methods.

Table 13. Comparison of GWNO3-BACKCAST nitrate load predictions to mass balance and soil nitrate sampling residual estimates.

Year	BACKCAST Model $NO3_{TotExcess}$ Mass Load Prediction (lbs/acre)[1]		
	Period A.15[2]	Period B.15[2]	Period A.15+B.15[2]
	Late Fall/ Early Winter[3]	Late Winter/ Early Spring[4]	Entire Wet Weather Season[5]
2004	146	84	230
2005	73	49	122
2006	44	50	94
2007	23	19	42
2008	42	42	84
Average	66	49	115

[1] Assumes 15% denitrification in the saturated zone. Value represents the nitrate mass load required to generate the observed groundwater nitrate-N concentrations, under the given model assumptions.
[2] See Appendix T for details.
[3] Values based on shallow groundwater nitrate-N mean observed during November 1 to December 31.
[4] Values based on shallow groundwater nitrate-N mean observed during January 1 to March 31.
[5] Values are the sum of columns A and B (leaching that occurred September 1 to March 31).

Estimates for $NO3_{TotExcess}$ may be biased low, because we assumed that there was no nitrate loss via denitrification in the unsaturated zone. In addition, the recharge term used in the calculation may be overestimated, because runoff of precipitation was assumed to be negligible.

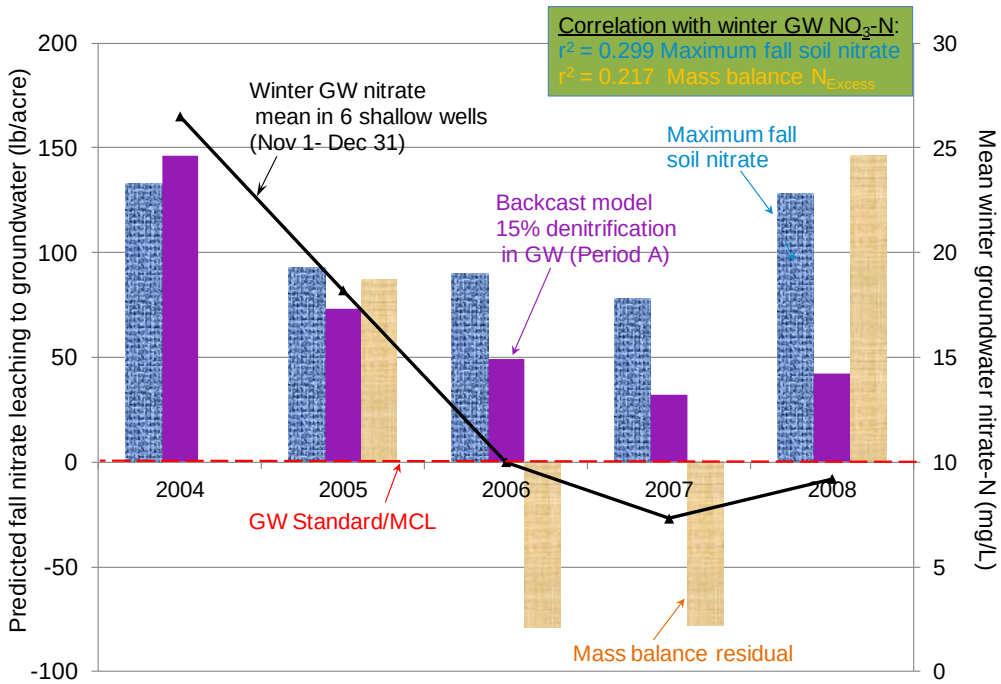

Figure 59. Comparison of Period A GWNO3-BACKCAST residual nitrate mass load predictions (purple bars) to residual mass estimates by mass balance (tan, light-textured bars) and soil nitrate results (blue, heavy-textured bars).

Key points revealed by the Period A model results include:

- $NO3_{TotExcess}$ calculated by the model for the late fall/early winter period was highest in 2004 and decreased through 2007, followed by a rebound in loading in 2008. This pattern is similar to the trend observed in the shallow groundwater nitrate concentrations (see Figure 59).
- The BACKCAST model calculated twice as much nitrate reaching the water table in the late fall/early winter of 2004 (146 lb/acre) as in the late fall/early winter of 2005 (73 lb/acre).
- The Method 1 mass balance approach estimated significant nitrogen deficits during both 2006 and 2007. The BACKCAST model, however, predicted excess nitrate mass loads occurring during the same period. A mass balance mean for multiple years (i.e., the current year and 2 previous years) may produce a result closer to the actual amount of nitrate leached and closer to the BACKCAST model results. Soil sample-derived residuals for 2006 and 2007 also indicated substantial excess nitrogen present in the soil column at the end of the growing season.
- In 4 out of 5 years, the residual nitrate mass estimated by maximum fall soil nitrate concentration was greater than the mass required by the BACKCAST model to generate the observed Period A groundwater nitrate condition. It is possible that the BACKCAST model assumptions were too conservative or that the maximum fall soil nitrate value does not represent the amount of nitrate reaching the water table due to factors such as denitrification in the soil or crop uptake after soil nitrate measurements.

- In 2008, both the mass balance and soil nitrate residual estimates indicate significantly more residual nitrogen mass was present in the soil column at the end of the growing season than the BACKCAST model $NO3_{TotExcess}$ predictions. Reasons for this discrepancy could be that the mass balance mineralization factor used was not accurate, the BACKCAST model assumptions were too conservative, or that denitrification in the soil removed a significant portion of the residual nitrate before it reached the water table.

GWNO3-BACKCAST predictions for late winter/early spring nitrate loading (Period B)

The BACKCAST model was also used to evaluate nitrate loading to the aquifer during the late winter/early spring period. This analysis was conducted to assess potential nitrate loading from ongoing mineralization beyond the fall months. For this time frame (Period B), the model was used to estimate the nitrate mass load required to generate the average shallow groundwater nitrate concentrations observed from January 1 to March 31 (compilation of shallow monitoring well data only). Similar to the Period A model analysis, a 15% saturated zone denitrification rate was assumed for the Period B evaluation.

The predicted Period B $NO3_{TotExcess}$ values are presented in Table 13. The nitrate mass loading predicted during Period B ranged from 19 to 84 lbs/acre, with an overall average of 49 lbs/acre.

Predicted nitrate loading for the entire wet-weather period (Period A+B)

The nitrate loading predicted by the BACKCAST model during Period B is *in addition to* the loading predicted to occur during Period A. To evaluate the total predicted wet-season nitrate load occurring at the site, the Period A and Period B $NO3_{TotExcess}$ values were summed for each year (Table 13; Figure 60). The nitrate mass load predictions for the entire high-recharge season (September 1 to March 31) ranged from 42 to 230 lbs/acre. The annual average $NO3_{TotExcess}$ predicted for the combined time frames was 115 lb/acre.

On average, the BACKCAST model predicted that approximately 43% of the total wet-season nitrate loading occurred between January 1 and March 31. In 2 years (2006 and 2008), the loading predicted to occur during the late winter and early spring was about the same as the loading predicted to occur during the late fall/early winter.

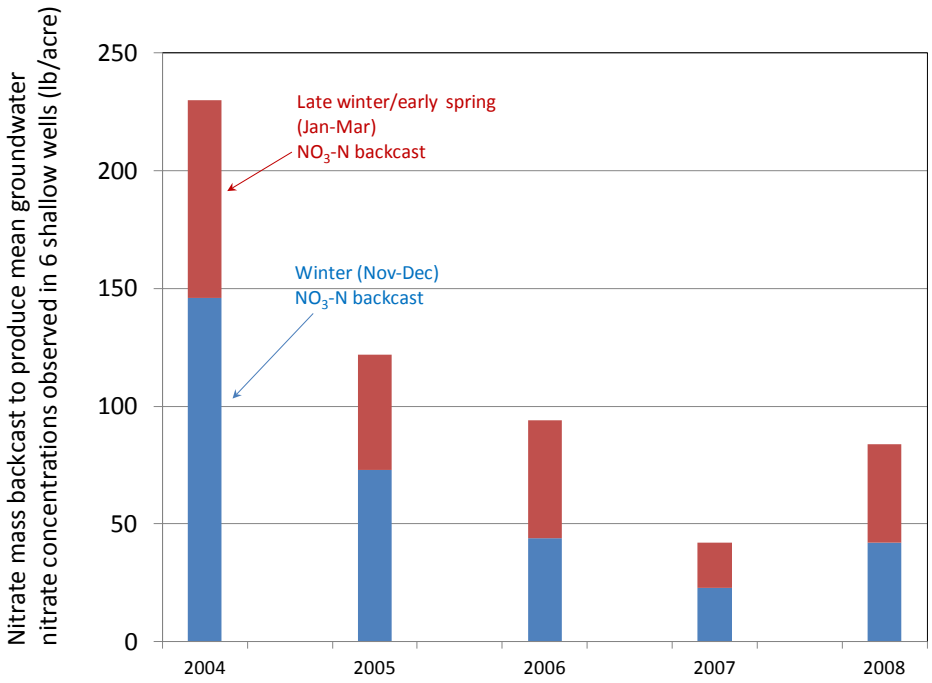

Figure 60. Total wet-season nitrate mass load predicted using the GWNO3-BACKCAST model.

Modeling implications

The BACKCAST loading estimates are approximations based on a variety of simplifying assumptions about nitrate transport and fate in the saturated zone. However, in most cases input values for the model variables were drawn directly from study data (e.g., background groundwater nitrate concentration, recharge rate, saturated zone hydraulic conductivity and hydraulic gradient, denitrification rate). The use of site-specific values, along with conservative modeling assumptions (for example, the saturated mixing zone was assumed to be only 2 feet thick), indicates that the modeled nitrate loading predictions are likely lower-bound values.

The mass balance and soil nitrate sampling approaches used in the study were intensive and more sophisticated than those typically used by dairy producers in Washington, yet these approaches were unreliable tools for predicting underlying groundwater nitrate conditions. The mass balance approach is limited by its reliance on variables that can be difficult to accurately quantify (especially internal loading contributions from mineralization). On the other hand, the study results show that the traditional post-harvest soil nitrate sampling approach can produce extreme variability in estimates of residual nitrate, depending on the time of collection. Soil sampling may also miss significant nitrate loss that occurs before samples are collected. Neither technique is well suited to account for mineralization that occurs throughout the latter half of the high-recharge season (January-March), which our analysis indicates can result in a substantial additional nitrate load to groundwater.

The poor correlation of the mass balance and soil sampling residual estimates with underlying groundwater conditions, and the BACKCAST modeling results that frequently suggest mass loading well in excess of these estimates, indicates that these techniques alone are not effective

tools for managing nutrients in a manner that is reliably protective of groundwater conditions. Direct monitoring of water quality at the water table remains the most accurate and reliable method for tracking impacts of manure management on groundwater.

Soil nitrate as an indicator of leaching to groundwater

What does the recommended fall soil nitrate guideline value mean for groundwater nitrate?

The post-harvest soil nitrate test (PSNT) target value, 15 mg/kg (Sullivan and Cogger, 2003), can be translated into a potential, although not highly reliable, amount of leachable nitrate when combined with seasonal recharge. This calculation is based on the fact that 27 lb of nitrogen, when mixed with 1 acre-foot of water, is equivalent to the groundwater MCL and Washington State groundwater quality standard of 10 mg/L-N of nitrate[10] (Harter, 2012). As mentioned previously, nitrate measured in the soil at 1-foot depth indicates only the amount remaining at that point in time and is typically considered an underestimate of the total amount actually leaching to groundwater (Harter, 2013; Zebarth, 2013; Osnoy et al., 2005). The amount already leached below 1 foot cannot be accurately estimated. Therefore, the fall soil nitrate values measured during the study (1-foot depth) cannot alone account for the entire residual nitrate in the soil at the end of the growing season that is likely to leach to groundwater.

Equation 20 shows a method for calculating the average hypothetical concentration of nitrate available for leaching to groundwater based on the amount of nitrogen available at the end of the growing season and the amount of fall recharge.

$$L_{NO3\text{-}N}=\frac{(RN_{Soil})}{(2.719)R_{Fall}} \qquad (Eq.\ 20)$$

where:
$L_{NO3\text{-}N}$ = Estimated leachate nitrate concentration (mg/L-N)
RN_{Soil} = Excess nitrogen as fall soil nitrate (lb/acre)
R_{Fall} = Fall/early winter recharge (September through December) (feet)
2.719 = Units conversion factor

This method assumes that all of the RN_{Soil} mixes with all of the R_{Fall} and is transported to the water table at one time with no additional nitrate subsequently added. This is, at best, a lower-bound assumption, because RN_{Soil} was highly variable over a weekly period even when duplicate samples were analyzed during the study, and there is no way to estimate the amount of nitrate that leached below the sampling zone between the soil samples (Viers et al., 2012; Zebarth, 2013).

Although Equation 20 provides a conservative estimate of the leachate concentration during the fall/early winter season, leaching can occur anytime, especially when precipitation exceeds evapotranspiration. Mineralization continues during and after the fall/early winter period, generating additional nitrate available for leaching in addition to the end-of-growing-season RN_{Soil} value (Zebarth, 2013; Zhao et al., 2010). Nitrate mineralized during the late winter/early spring

[10] Chapter 173-200 WAC

has been found to quickly leach to the water table in the nearby Abbotsford, B.C. area (Zebarth, 2013).

Table 14 shows the results of combining the estimated R_{Fall} values (Appendix J, Table J.5) with the target PSNT guideline for grass of 15 mg/kg (Sullivan and Cogger, 2003; 47 lbs/acre for the study site) using Equation 20 and assuming no additional nitrate in the second foot of soil. The range of calculated leachate nitrate concentrations for the site is 11 to 18 mg/L-N, which exceeds the Washington State groundwater quality standard for nitrate[11].

Table 14. Estimated nitrate-N concentration in fall/early winter leachate assuming that 15 mg/kg of soil nitrate in the top 1 foot at the study site is mixed with the fall/early winter recharge during the study.

	2004	2005	2006	2007	2008	Average
Fall recharge, R_{Fall} (feet)[1]	1.57	1.19	1.39	0.98	1.32	1.29
Nitrate-N concentration in water with 15 mg/kg soil nitrate mixed with R_{Fall} *(bulk density conversion= 3.1)*	11	14	12	18	13	14

[1] September 1 through December 31.

The amount of residual fall soil nitrate that, when mixed with the average observed fall recharge, 1.3 feet, would be equivalent to 10 mg/L is 35 lb/acre (11 mg/kg). This is a small amount of nitrate relative to the accuracy of the PSNT.

Fall soil nitrate variability and sample timing

Results for PSNT sampling can be highly variable during the period recommended for sampling (after the last harvest and before 5 inches of fall precipitation; Figure 26, Plate 2). Factors such as timing of manure application, temperature, the amount and timing of recent precipitation, and inherent heterogeneity of nitrate in soil all affect the concentration of nitrate in a soil sample in the fall on a given day (Sullivan and Cogger, 2003; Oenema et al., 2010).

The range for weekly fall soil nitrate concentrations from September 1 to October 31 each year was 12 to 43 mg/kg (37 to 133 lb/acre; Table 15). The large variation in soil nitrate concentrations during the fall in the 0 to 1-foot depth interval illustrates that a single sample is unlikely to correspond with the actual total amount of nitrate leaching to groundwater.

Table 15 also shows the results of combining R_{Fall} values from Table 14 with either the minimum or maximum fall soil nitrate concentrations using Equation 20. The annual range in variation between the highest and lowest soil nitrate-derived leachate concentration was 12 to 18 mg/L-N.

[11] Chapter 173-200 WAC

Table 15. Fall soil nitrate variability and resulting soil nitrate-derived leachate concentrations for soil samples collected from September 1 through October 31.

Year	Minimum soil nitrate (mg/kg)	Maximum soil nitrate (mg/kg)	Difference between minimum and maximum soil nitrate (mg/kg)	Fall recharge (feet)	Minimum estimated leachate nitrate concentration (mg/L-N)[1]	Maximum estimated leachate nitrate concentration (mg/L-N)[1]	Difference between minimum and maximum leachate estimates (mg/L-N)
2004	19	43	24	1.57	14	31	18
2005	12	30	19	1.19	11	29	18
2006	16	29	14	1.34	13	25	12
2007	12	25	14	0.98	14	30	16
2008	21	41	20	1.32	18	36	18

[1] This assumes that recharge mixes completely with the soil nitrate and that 27 lb/acre of nitrogen in 1 acre-foot is equivalent to 10 mg/L nitrate-N.

Reasons for high seasonal variability in fall soil nitrate measurements include combinations of the following:

- Inherently high spatial variability of nitrate in soil and over short time periods.
- Timing of the last harvest (which requires a period of dry weather).
- Re-wetting of soil in the fall after drying all summer, which causes a surge in mineralization; percolating water carries recently mineralized nitrate to groundwater.
- Timing of the last manure application (sometimes after the last harvest).
- Crop uptake and removal, which depends on complex interactions between temperature, soil moisture, precipitation, and irrigation.

In most years, when the maximum fall soil nitrate concentration substantially exceeded 15 mg/kg (2004, 2006, and 2008), winter groundwater nitrate concentrations increased in most, if not all, monitoring wells (Figure 40, Plate 5). In contrast, during the winter following the lowest maximum fall soil nitrate concentration (2007), nitrate decreased in some shallow wells, and increases in other wells were lower than in other years.

Estimated leachate nitrate concentrations based on mean and maximum fall soil nitrate concentration

If the average 1-foot fall soil nitrate concentration for each year were mixed with the recharge for the season (Equation 20), the predicted leachate nitrate concentration would be 16 to 23 mg/L-N as shown in Table 16. If the maximum fall soil nitrate concentration were combined with wet-season recharge, the annual predicted leachate nitrate concentration would be 29 to 50 mg/L-N. (These calculated concentrations include soil nitrate values from November 1-15, 2 weeks more than the previous fall soil nitrate analyses, to account for any nitrate that might have affected groundwater after the normal soil sampling period.)

Table 16. Estimated leachate concentration if the fall soil nitrate at 1-foot depth during the study (mean or maximum from September 1 - November 15) were mixed with the fall/early winter (September-December) recharge observed.

	2004	2005	2006	2007	2008
Fall/early winter recharge (feet)[1]	1.57	1.19	1.39	0.98	1.32
Mean fall soil nitrate (lb/acre)[2]	82	52	77	48	81
Estimated leachate nitrate (mg/L-N)[3]-mean	**19**	**16**	**21**	**18**	**23**
Maximum fall soil nitrate (lb/acre)[2]	133	93	186	78	128
Estimated leachate nitrate-N (mg/L)-max	**31**	**29**	**50**	**29**	**36**

[1] September 1 through December 31
[2] September 1 through November 15
[3] Based on Equation 20

Conclusions

Intensive monitoring of soil, manure, crop, and groundwater showed that management practices at a manured dairy grass field over the Sumas-Blaine Aquifer (SBA) resulted in mean shallow groundwater nitrate concentrations of 5.5 to 30 mg/L-N. Fifty-six percent of monthly mean groundwater nitrate results were above 10 mg/L-N. Mean groundwater nitrate results were below 10 mg/L-N following application of manure nitrogen at rates close to crop removal rates and at appropriate times.

Nitrate concentrations in shallow groundwater underlying the study field generally declined over the first 3 years of the study (2005-2007) from an average concentration in 6 shallow monitoring wells of 30 mg/L-N (maximum of 45 mg/L-N). This decline is interpreted to be primarily the result of a steady reduction in the amount of nitrogen applied to the field. In the fourth year (2008), groundwater nitrate concentrations increased presumably due to an excess of manure nitrogen inputs relative to outputs. The annual mass of nitrogen removed in the crop was consistent throughout the study and did not correspond with the large variation in the annual mass of nitrogen applied (394-715 lb/acre).

Factors affecting nitrate levels in groundwater and soil

Many factors affected groundwater and soil nitrate concentrations relative to manure applications. Factors related to environmental and management practices act simultaneously and in complex ways. These factors resulted in high spatial and temporal variability in groundwater and soil conditions.

The main environmental factors that influenced conditions in the study include:

- Hydrogeologic conditions
  - The shallow groundwater elevation in winter led to rapid transport of nitrate to groundwater.
  - We found evidence of substantial loss of nitrate to nitrogen gas in 4 out of 6 monitoring wells via microbially-mediated denitrification. Denitrification is less likely to occur in coarse-grained materials.
- Precipitation
  - Most of the annual precipitation in the area occurs during the period of limited crop growth (October through March). Precipitation that infiltrated through the soil to the groundwater (recharge) carried available nitrate to the water table.
- Soil conditions

  Summertime drying, which we observed in the beginning of the study, may have slowed mineralization and dampened crop uptake. This in turn may have increased the amount of residual nitrate available to leach to groundwater in the fall.

Management practices that influenced conditions in the study include:

- Rate of nitrogen application (external loading): The more nitrogen applied to the field in excess of the crop demand, the higher the amount of nitrate that reached the water table. Higher shallow groundwater nitrate concentrations occurred in the early winter following nitrogen application in excess of crop uptake (2005, 2008). Groundwater nitrate concentrations were lower (and mostly below 10 mg/L-N) when the amount of nitrogen applied was less than crop removal (2006, 2007).
- Rate of nitrogen mineralization (internal loading): Organic nitrogen in the soil from previous years, and manure organic nitrogen from the current year, are bacterially converted to plant-available nitrate, but accurately quantifying the amount and timing is difficult. The average estimated annual amount of mineralized nitrogen available at our study site was roughly one-half of the annual amount of plant-available nitrogen. Mineralization contributed to groundwater nitrate loading throughout the high-precipitation season.
- Tillage effects: Tillage of the field during the first year (2004) led to elevated nitrogen mineralization in the soil. This contributed to groundwater nitrate loading at the beginning of the study but was not the dominant factor.
- Timing of manure applications:
  - Manure application late in the growing season (or after the growing season), when recharge was high, was followed by elevated groundwater nitrate concentrations.
  - Manure application too early in the spring was followed by elevated groundwater nitrate concentrations.
- Denitrification: Denitrification in shallow groundwater at the site was controlled by reduced dissolved oxygen concentrations in 4 out of 6 wells. We estimated that an average of 28% of the nitrate in 4 wells with at least occasionally low dissolved oxygen levels was converted to nitrogen gas. Denitrification was more prevalent in the summer and fall, when recharge was low, than in the winter.

Nitrogen residual estimates compared with groundwater nitrate concentrations

An important goal of the Dairy Nutrient Management Plans (DNMPs) is to minimize the amount of residual nitrogen left in the field after the growing season and available to leach to groundwater.

Mass balance and post-harvest soil nitrate residual estimates

We evaluated 2 field-based methods for estimating the amount of residual nitrogen left in the study field at the end of each growing season:

- Mass balance estimate of annual nitrogen residual
- Fall soil nitrate residual (1 foot depth)

The mass balance analysis used the most recent Oregon State University Extension Service method for estimating plant-available nitrogen. The annual nitrogen residual from 2005 to 2008

was -79 lb/acre to 146 lb/acre with a 4-year average of 19 lb/acre. Significant nitrogen deficits were calculated for 2006 and 2007. The annual mass balance residual did not correspond well with the nitrogen application rates. The main source of uncertainty in the mass balance evaluation was probably the contribution from mineralized organic matter from past years.

The second method used the fall soil nitrate values collected weekly in September and October to estimate the annual mean and maximum soil nitrate residual. The 4-year average fall soil nitrate result was 72 lb/acre; the maximum was 104 lb/acre. These averages were 4 to 5 times the average calculated by the mass balance method, and have high variability over short time periods. Soil nitrate residual values did not track closely with nitrogen application rates.

Fall soil nitrate results are an underestimate of the amount of nitrate available for leaching at the end of the growing season, because additional nitrate in the soil at 2 feet or deeper was not included in the calculation as well as any nitrate that leached before or after the sample was collected.

If the average 1-foot deep fall soil nitrate concentration for each year were mixed with the recharge for the fall/early winter season, the predicted leachate nitrate concentration would be 16 to 23 mg/L-N. If the maximum fall soil nitrate concentration for each year were combined with recharge, the annual predicted leachate nitrate concentration would 29 to 50 mg/L-N.

We attempted to correlate results of these 2 nitrate residual estimate methods with early winter groundwater nitrate concentrations, because early winter is typically the time when most of the end-of-season residual nitrate in the soil reaches the water table. Neither approach correlated well with groundwater nitrate concentrations that we observed in our study.

The nitrogen mass balance method contains many variables with inherent uncertainty, especially nitrate mineralized from past manure applications. This high uncertainty may explain the lack of correlation with groundwater concentrations. The high variability of soil nitrate conditions during the post-harvest soil sampling period likewise contributed to the unreliability of the soil nitrate method for predicting early winter groundwater nitrate concentrations. In addition, neither the nitrogen mass balance method nor the fall soil nitrate method takes into account processes that occur in the groundwater, such as mixing and denitrification.

GWNO3-BACKCAST model

An alternative method for estimating residual nitrate, the GWNO3-BACKCAST model, predicted the amount of nitrate needed to create the groundwater nitrate concentrations observed. We used this method, because the mass balance and soil nitrate methods were not good predictors of nitrate leaching. The model is based mainly on parameters measured during the study: recharge rate, groundwater nitrate, hydraulic conductivity and horizontal gradient.

In this study, on average the BACKCAST model predicted an average nitrate loading to groundwater during the wet season (September-March) of 115 lb/acre, with a range of 42 to 230 lb/acre. Of the total predicted nitrate loading to groundwater, 66 lb/acre occurred during the late fall/early winter and 49 lb/acre during the late winter/early spring. The BACKCAST model

predictions were most sensitive to the recharge rate, the denitrification rate in groundwater, and the thickness of the groundwater mixing zone.

Post-harvest soil nitrate test

We evaluated the recommended post-harvest soil nitrate concentration for dairy grass fields in western Washington, 15 mg/kg, in relation to the recharge amounts observed. The calculated leachate nitrate concentration that would result from combining this soil nitrate concentration with the observed fall/early winter recharge volumes ranged from 11 to 18 mg/L-N. This does not include nitrate available for leaching below the top foot of soil. All soil nitrate samples collected according to the recommended protocols for post-harvest soil nitrate testing during the study exceeded 15 mg/kg.

If the seasonal *average* September to mid-November soil nitrate concentrations was used in the same calculations with the recharge that occurred during the study, the estimated nitrate concentration in leachate would have been 16 to 23 mg/L-N. If the seasonal *maximum* September to mid-November soil nitrate concentration was used in the calculation, the estimated leachate nitrate concentration would have been 29 to 50 mg/L-N.

Recommendations

Reducing nitrate leaching to groundwater at manured dairy fields over the Sumas Blaine Aquifer (SBA), like the one in this study, will require improving manure application based on evolving science and technology. This includes fine-tuning nitrogen loading analyses and taking groundwater into account. Groundwater monitoring will be needed to evaluate the effectiveness of measures to reduce nitrate loading to groundwater.

Based on the results of this study, the following actions are recommended to promote improvements to groundwater quality in the SBA and in other areas of Washington State with similar conditions.

Reduce nitrate loading to groundwater

We recommend that stakeholders develop a process whereby (1) manure and fertilizer nitrogen inputs and outputs are tracked on a field-by-field basis and (2) the information is used to minimize nitrate leaching below the root zone. Because of high seasonal rainfall and shallow depth to water, appropriate timing and amount of nutrient application is crucial. Involvement of state and local organizations, in partnership with universities, dairy and other agricultural producers, is needed to improve nitrogen use efficiency and to protect groundwater quality.

Key challenges for local conditions include:

- Availability of management practices to prevent nitrate leaching.
- Knowledge of management practices to prevent nitrate leaching.
- Part of the growing season overlaps with the groundwater recharge season.

Lessons learned in this study that could decrease nitrate leaching to groundwater in the SBA include:

- Review available mass balance assessment methods for use in Dairy Nutrient Management Plans (DNMPs). Develop or adapt existing methods so they more accurately account for effects on groundwater (i.e., more accurate assessment of soil organic matter contribution).

- Calculate nitrogen applications (manure, fertilizer, irrigation water) and crop removal based on measured amounts and nitrogen analyses. This is especially important in areas where groundwater nitrate already does not meet the drinking water standard. Timing of nitrogen application relative to recharge is especially important.

- Our results indicate that it is best to schedule the last manure application by late August to early September. Manure application during the high-recharge period (September through mid-March) is likely to increase nitrate leaching to groundwater.

- Where groundwater is well-oxygenated and denitrification rates are low, take special care to prevent overapplication or application during the high-recharge period.

- If soil moisture is low in the summer, consider irrigating to increase mineralization and nitrification and increase available nitrate for the crop. Avoid overapplication of irrigation water to prevent nitrate leaching.
- Consider extending the time between tillage events to decrease the amount of nitrogen reaching groundwater.
- Consider limiting manure application to forage crops during the first season following tillage.
- Consider updating the post-harvest soil nitrate test (PSNT) guidance to incorporate hydrologic influences (i.e., local aquifer recharge) based on the expertise of land grant universities as well as local and state scientists.
- Until revised, use the existing post-harvest soil nitrate protocols (methods and timing) and targets (15 mg/kg for grass; 20 mg/kg for corn) as criteria for evaluating manure management at dairy operations in western Washington.

Monitor to evaluate the effectiveness of management improvements

A program is needed to determine how well current and future manure management practices are working to improve groundwater quality. Because there is no reliable substitute, direct groundwater monitoring using dedicated monitoring wells is a key component of an effectiveness monitoring program.

Although groundwater monitoring is the only available way to determine the amount or the concentration of nitrate that actually reaches the water table, soil nitrate monitoring in the fall is a necessary tool for on-farm nutrient management. If conducted with limitations in mind, soil nitrate monitoring also can serve as a screening tool for closer inspection of groundwater conditions.

Nitrogen mass balance evaluations are also an important tool for dairy producers to manage nutrients and identify potential courses of action to address high soil and groundwater nitrate concentrations. Current methods for analyzing mass balances for DNMPs should be evaluated to ensure the methods are as accurate as possible.

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Appendices

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Appendix A. Glossary, Acronyms, and Abbreviations

Glossary

Aerobic: In terms of liquid water, the state that contains oxygen at a measurable level.

Ammonium: A positively charged ion that is available to plants and typically comprises a large portion of fresh manures. The symbol for ammonium is NH_4^+

Anaerobic: In terms of liquid water, devoid of oxygen at a measurable level.

Cation: A positively charged ion.

Coefficient of curvature: A number used to classify materials as sands or gravels based on the particle size distribution of a soil sample. The number is based on the particle diameters corresponding to 10, 30, and 60%, respectively passing on a standard cumulative particle size distribution curve.

Denitrification: The bacterial or chemical process whereby nitrate is converted to nitrogen gas, usually in a reducing/low dissolved oxygen environment.

Depth to water: The distance from a measuring point (i.e., the ground surface) to the saturated zone, usually measured in a well or piezometer.

Effective grain size: Also called effective diameter, d_{10}, is the particle diameter corresponding to 10% finer on the grain-size curve.

Evapotranspiration: The sum of evaporation and transpiration. Evaporation is the loss of water to the atmosphere from the ground surface down to the capillary fringe of the water table. Transpiration is the evaporation of water from plant leaves.

Groundwater: Water in the saturated zone that is under pressure that is equal to or greater than atmospheric pressure.

Hydraulic conductivity (K_H): Rate at which water moves through a material at a unit gradient and depends on the size and arrangement of the pores between the particles.

Hydrogeologic: Relating to the scientific study of the waters of the earth, especially with relation to the effects of precipitation and evaporation upon the occurrence and character of water in streams, lakes, and on or below the land surface.

Leachate: Water that percolates through a porous media carrying dissolved substances as it moves, typically to the groundwater.

Mass balance: A tool for estimating nutrient budgets that include inputs, outputs, and residuals for a system (e.g., nitrogen in an agricultural field).

Maximum contaminant level (MCL): A regulatory limit set by the U.S. Environmental Protection Agency (EPA) for contaminants in drinking water. If an MCL is exceeded, regulatory action is required under the Safe Drinking Water Act.

Methemoglobinemia: A serious health condition that reduces the ability of red blood cells to carry oxygen that can result from exposure to high levels of nitrate. In most adults and children, these red blood cells rapidly return to normal. However, in infants it can take much longer for the blood cells to return to normal. Infants who drink water with high levels of nitrate (or eat foods made with nitrate-contaminated water) may develop a serious health condition due to the lack of oxygen and call "blue-baby syndrome."

Mixing zone: The top portion of an aquifer into which recharging water is mixed over a specified period of time.

Monitoring well: A cased hole drilled and completed to specifications that enable water level and water quality sampling to be conducted that are representative of conditions in the portion of the aquifer of interest.

Nitrate: A common, highly mobile, nitrogen-based chemical compound. Ammonium from manure is typically converted to nitrate by bacteria. The symbol for nitrate is NO_3^-

Nitrogen: An element that is found in all parts of the environment and in organic/living matter.

Nitrate-N: The common reference to nitrate as nitrogen in water analyses.

Nutrient: Substance such as carbon, nitrogen, and phosphorus used by organisms to live and grow. Too many nutrients in the water can promote algal blooms and rob the water of oxygen vital to aquatic organisms. Nutrients can be transported from groundwater to surface water. Nitrate is considered a nutrient.

Recharge (noun)**:** The amount of water entering the saturated zone at the water-table surface over a period of time.

Root zone: The soil zone in which crop roots are found. The thickness varies with the type of crop. The root zone typically contains both water and air and is unsaturated.

Specific capacity: A measure of the productivity of a well estimated by measuring the pumping rate (yield) and dividing by the change in the height of water in the well (drawdown)

Study period: September 22, 2004 through March 18, 2009.

Study site (also study field): 22-acre dairy located northwest of Lynden, Washington.

Unconfined aquifer: An aquifer containing water that is not under pressure; the water level in a well is the same as the water table outside the well.

Uniformity coefficient: A number used to classify materials as sands or gravels based on the particle size distribution of a soil sample. The ratio of d_{60} to d_{10} where d_{60} is the particle diameter

corresponding to 60% finer on the cumulative particle size-distribution curve, and d_{10} is the particle diameter corresponding to 10% finer on the cumulative particle size-distribution curve.

Upgradient: In hydrology, an *upgradient* location is one that exhibits a larger hydraulic head in comparison to a *downgradient* location. Water flows from areas of high hydraulic head to areas of low hydraulic head. Hydraulic head is the total pressure exerted by a water mass at any given point. Total hydraulic head is the sum of elevation head, pressure head, and velocity head.

Vadose zone: The subsurface zone that starts at the land surface and contains both air and water (is not saturated).

Water table: The level in the saturated zone at which hydraulic pressure is equal to atmospheric pressure and is represented by the water level in wells that are not pumping.

Acronyms and Abbreviations

B.C.	British Columbia, Canada
BGS	Below ground surface
DO	Dissolved oxygen
DNMP	Dairy Nutrient Management Plan
DOC	Dissolved organic carbon
Ecology	Washington State Department of Ecology
GPS	Global Positioning System
GW	Groundwater
K_H	(See Glossary above)
MCL	(See Glossary above)
MEL	Manchester Environmental Laboratory
N	Nitrogen
n	Number
NH_4^+	Ammonium
Nitrate-N	Nitrate as nitrogen
NO_3^-	Nitrate
NRCS	Natural Resources Conservation Service
PAN	Plant-available nitrogen
PSNT	Post-harvest soil nitrate test
PVC	Polyvinyl chloride
RCW	Revised Code of Washington
RPD	Relative percent difference
RSD	Relative standard deviation
SBA	Sumas-Blaine Aquifer
SOP	Standard operating procedure
TPN	Total persulfate nitrogen
U.S. EPA	U.S. Environmental Protection Agency
WAC	Washington Administrative Code
WSU	Washington State University

Units of Measurement

°C	degrees centigrade
dw	dry weight
ft	feet
in	inches
kg	kilograms, a unit of mass equal to 1,000 grams.
km	kilometer, a unit of length equal to 1,000 meters.
lb	pound
mm	millimeter
mg/kg	milligrams per kilogram (parts per million)
mg/L	milligrams per liter (parts per million)
mg/L-N	milligrams per liter as nitrogen
mL	milliliters
s.u.	standard units
ug/L	micrograms per liter (parts per billion)
um	micron
umhos/cm	micromhos per centimeter, a unit of conductivity

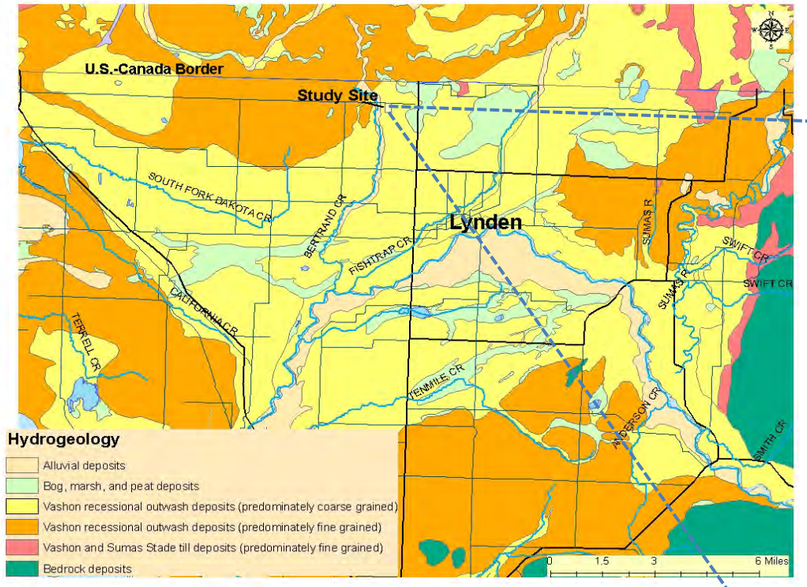
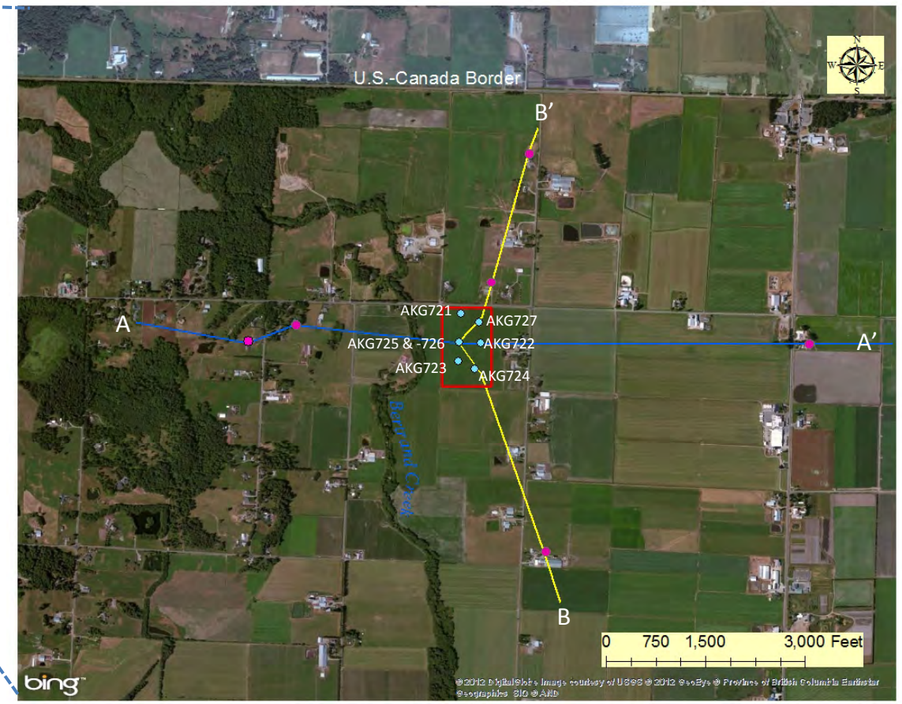


Figure 4. Local surficial geology of the study area from Jones (1999).

Figure 5. Location of wells used in Figure 6 cross-sections. The blue dots represent monitoring wells used in this study; pink dots private, domestic wells. The red line is the study area boundary.



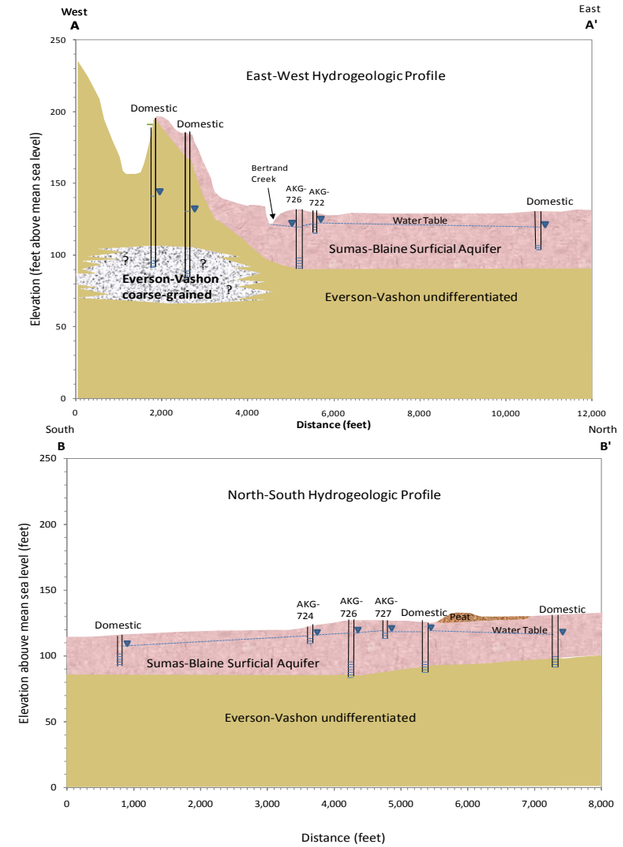


Figure 6. Generalized hydrogeologic cross-sections from Figure 5.

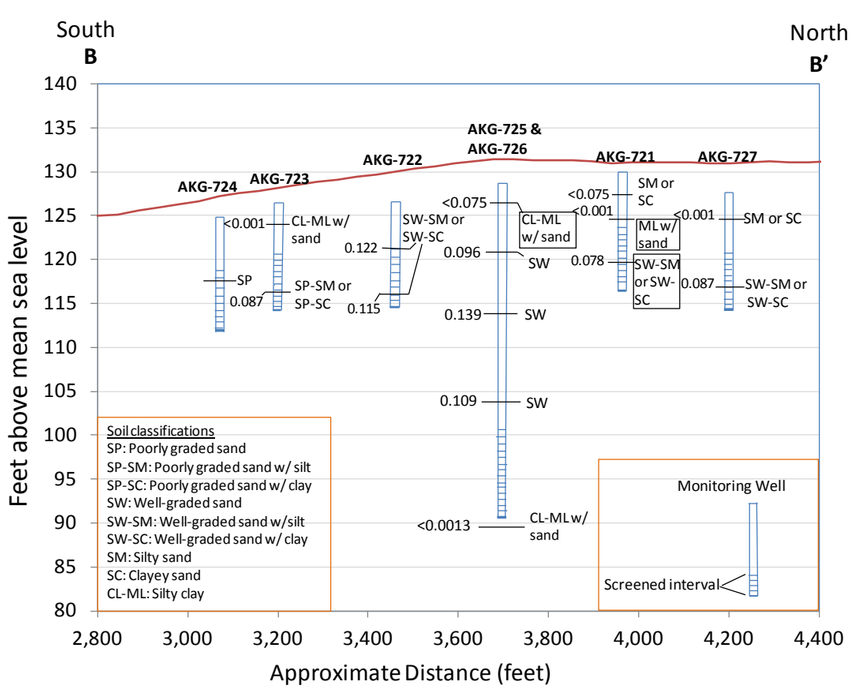


Figure 27. Soil classifications and d_{10} values for soil samples projected on land surface cross-section B-B' in Figure 5.

Plate 1. Hydrogeologic data

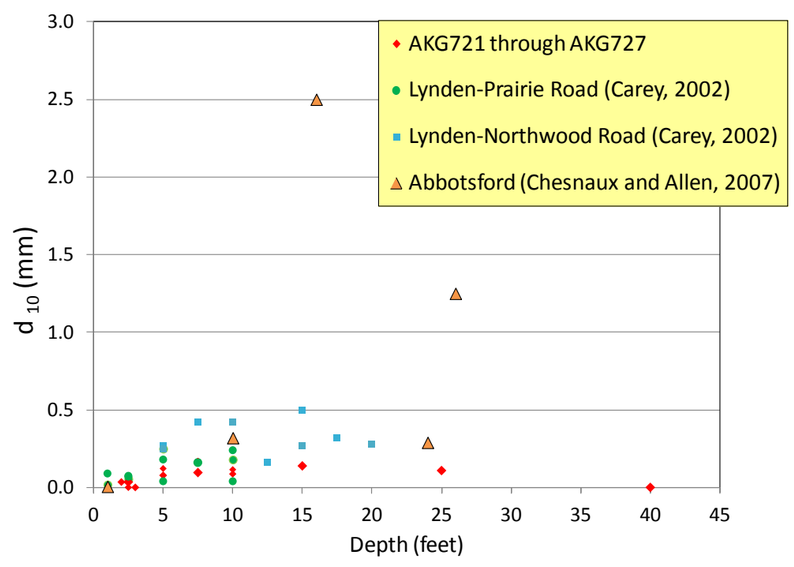


Figure 43. Effective grain size (d_{10}) values for drilling samples from monitoring wells in the U.S. and from one well near Abbotsford, B.C. See Figure 44 for locations.

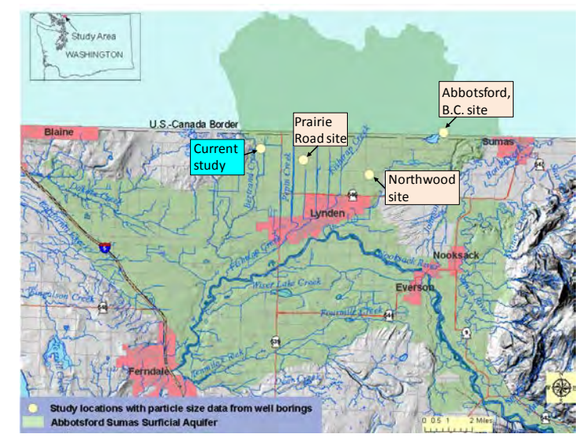


Figure 44. Locations of particle size analysis samples from well borings shown in Figure 43 (Chesnaux and Allen, 2007; Carey, 2002).

Soil Conditions

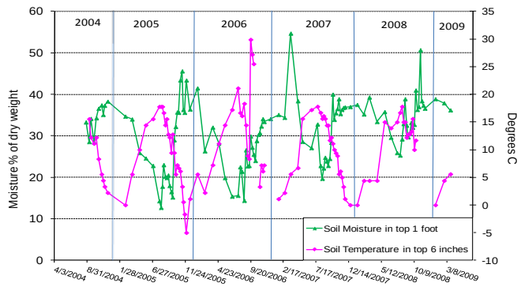


Figure 17. Soil moisture and soil temperature measurements.

Inputs of Nitrogen and Chloride

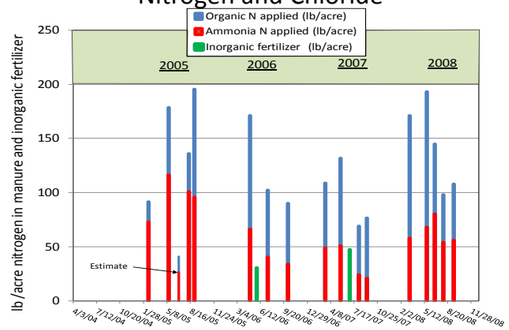


Figure 19. Total nitrogen applied in manure and inorganic fertilizer. Each bar represents one application event.

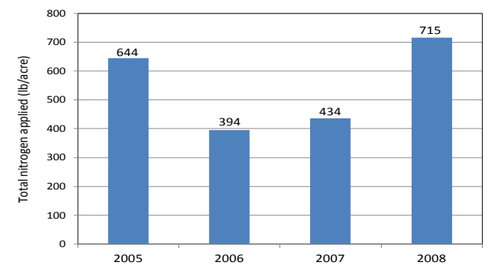


Figure 20. Total nitrogen applied annually (manure, fertilizer, and irrigation water).

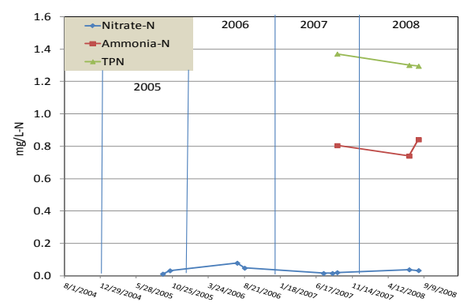


Figure 21. Concentrations of nitrate-N, ammonia-N, and total persulfate N in irrigation water applied.

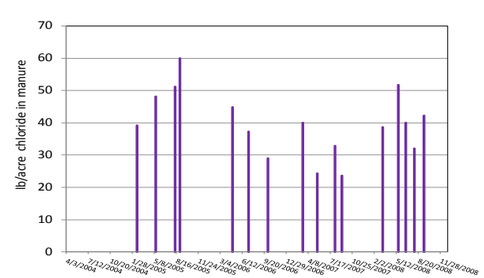


Figure 22. Chloride applied to the field in manure.

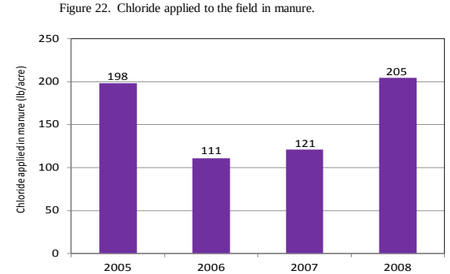


Figure 23. Chloride applied annually to the field in manure.

Outputs of Nitrogen

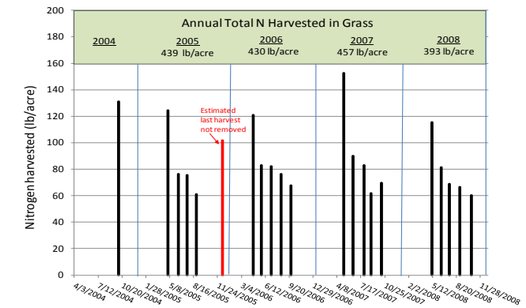


Figure 24. Total nitrogen harvested in grass for each cutting (individual bars) and annual totals for 2005 through 2008 (along the top) .

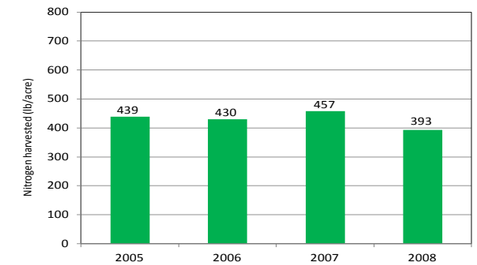


Figure 25. Total nitrogen removed in the grass harvest, including an estimate for uptake in 2005, which was not removed from the field.

Residual Nitrate

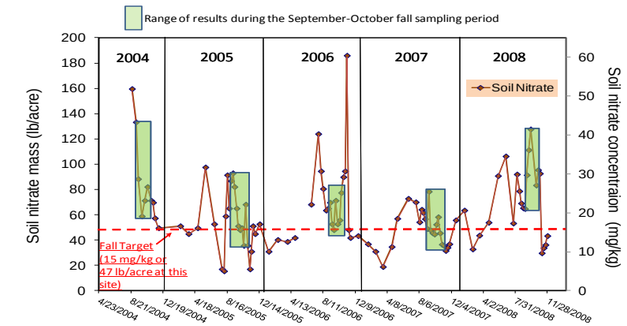


Figure 26. Soil nitrate results at 1-foot depth. Green shaded areas indicate results for the typical fall soil sampling time, September through October. Red line indicates the level below which management changes are not recommended (Sullivan and Cogger, 2003).

Plate 2. Soil conditions, nitrogen and chloride inputs, nitrogen outputs, and soil nitrogen residual.

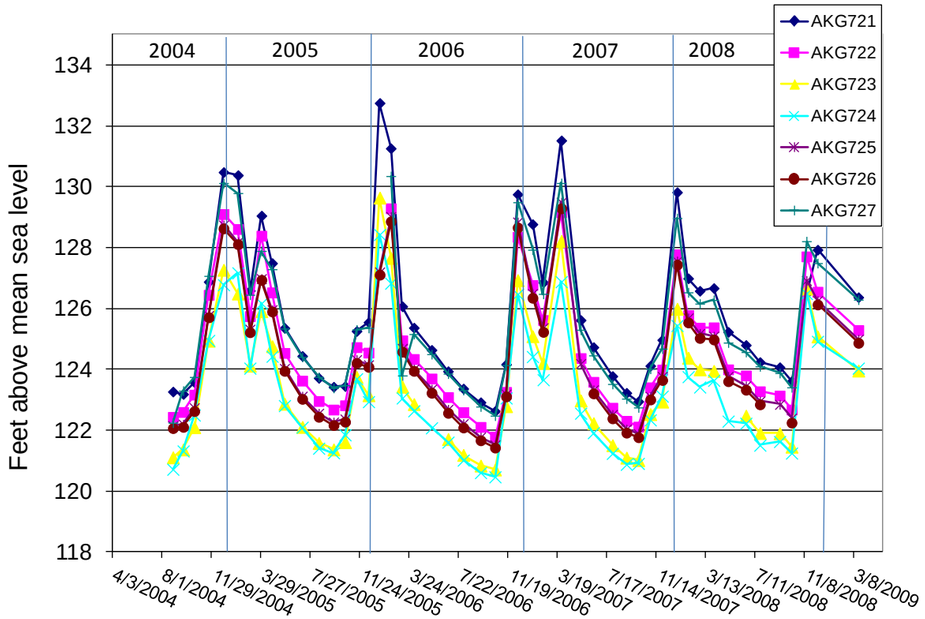


Figure 28. Water table elevations in the monitoring wells relative to the top of casing of AKG721 (134.00 feet, NAVD88) and depth to water below ground surface.

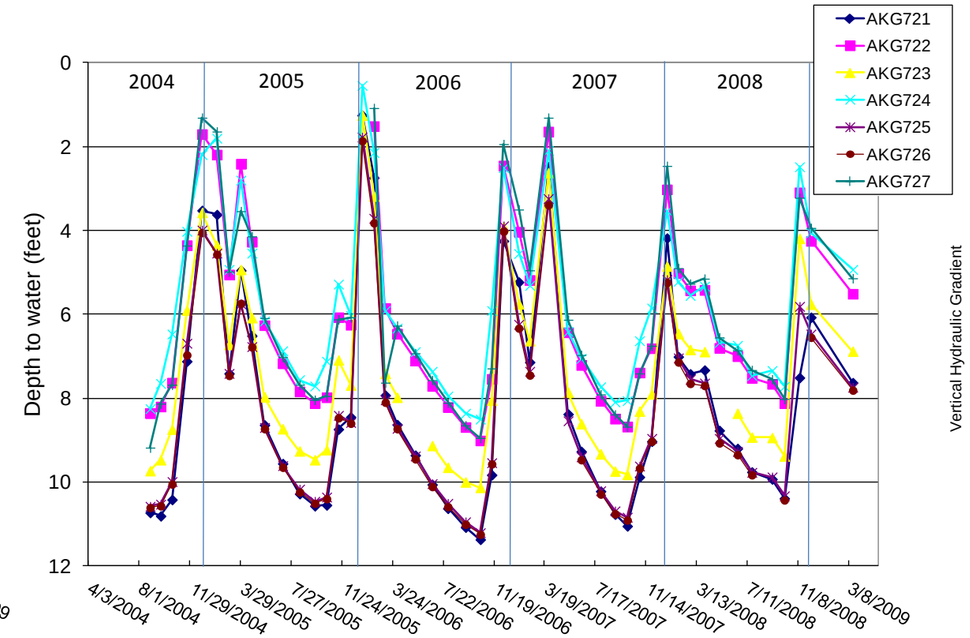


Figure 29. Depth to water below ground surface in the monitoring wells.

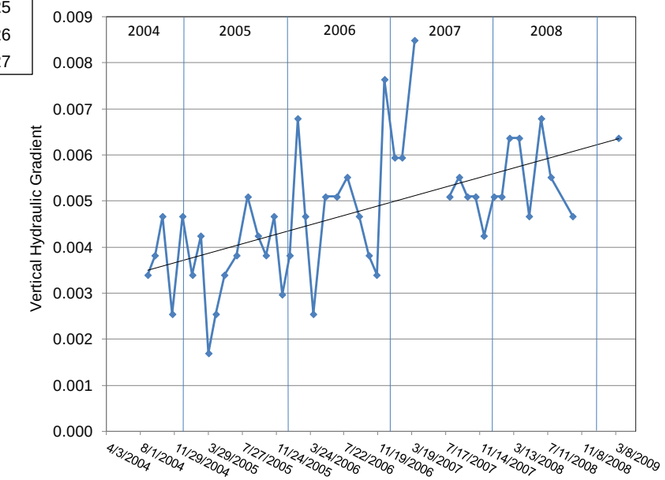


Figure 31. Downward vertical hydraulic gradient at side-by-side wells, AKG725 (13 feet deep) and AKG726 (38 feet deep).

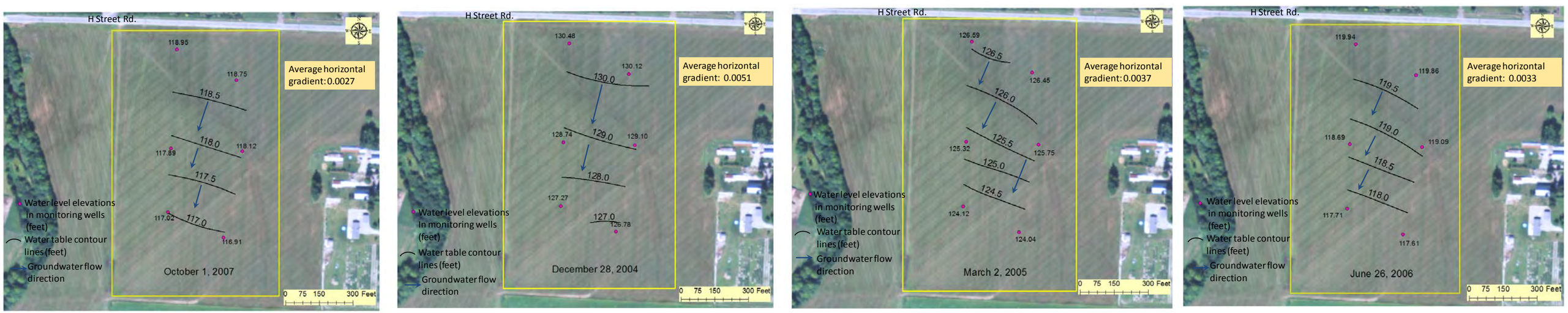


Figure 30. Water level contours in feet based on depth-to-water measurements on October 1, 2007 (fall), December 28, 2004 (winter), March 2, 2005 (spring), and June 26, 2006 (summer) and average horizontal gradient on each date. Elevations are relative to the top of casing at AKG721 (134.00 feet, NAVD88).

Plate 3. Water table elevations, depths, contours, and vertical hydraulic gradient.

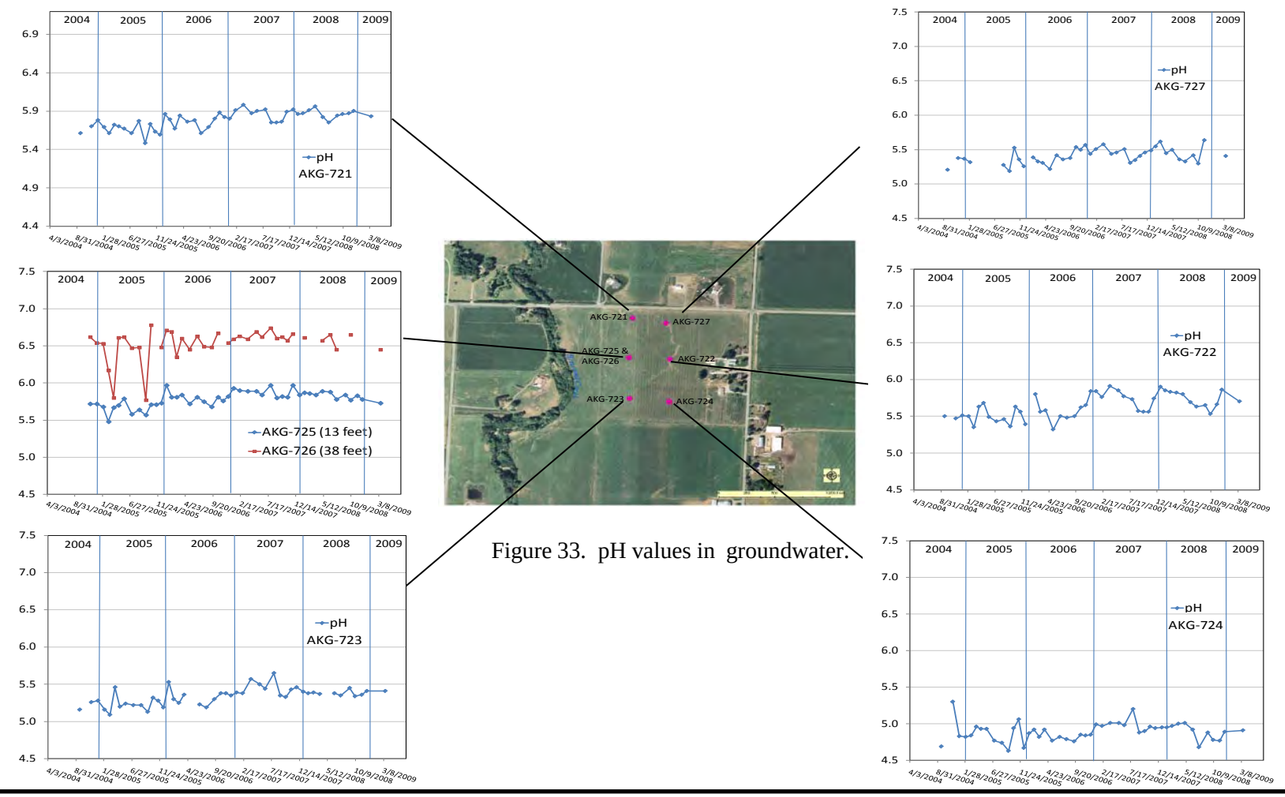


Figure 33. pH values in groundwater.

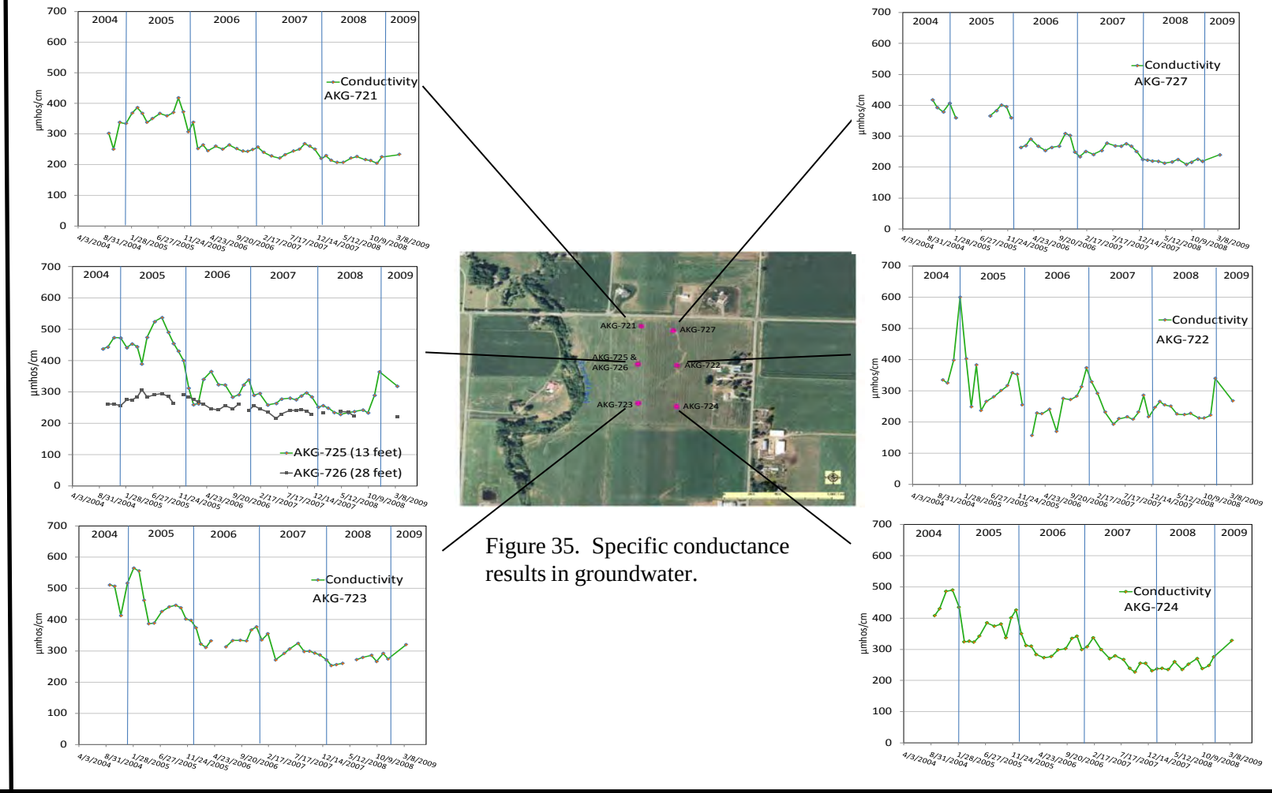


Figure 35. Specific conductance results in groundwater.

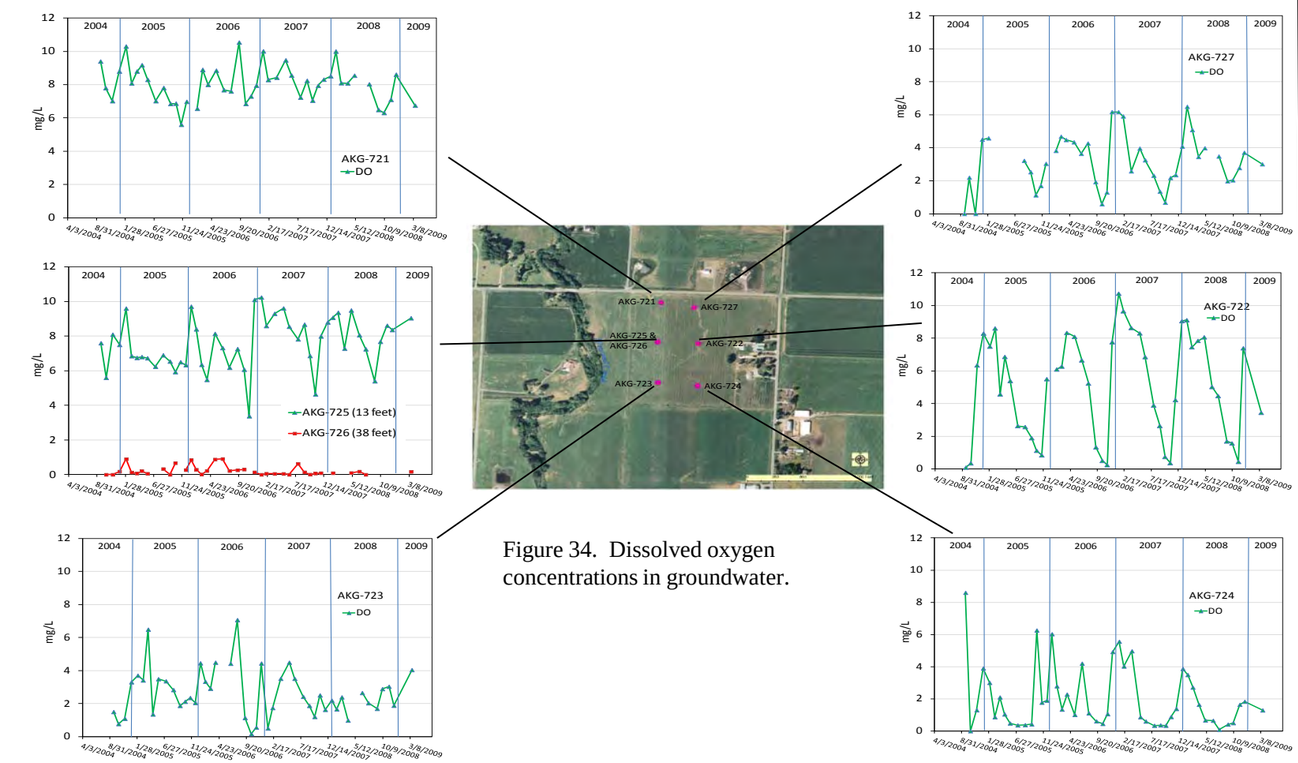


Figure 34. Dissolved oxygen concentrations in groundwater.

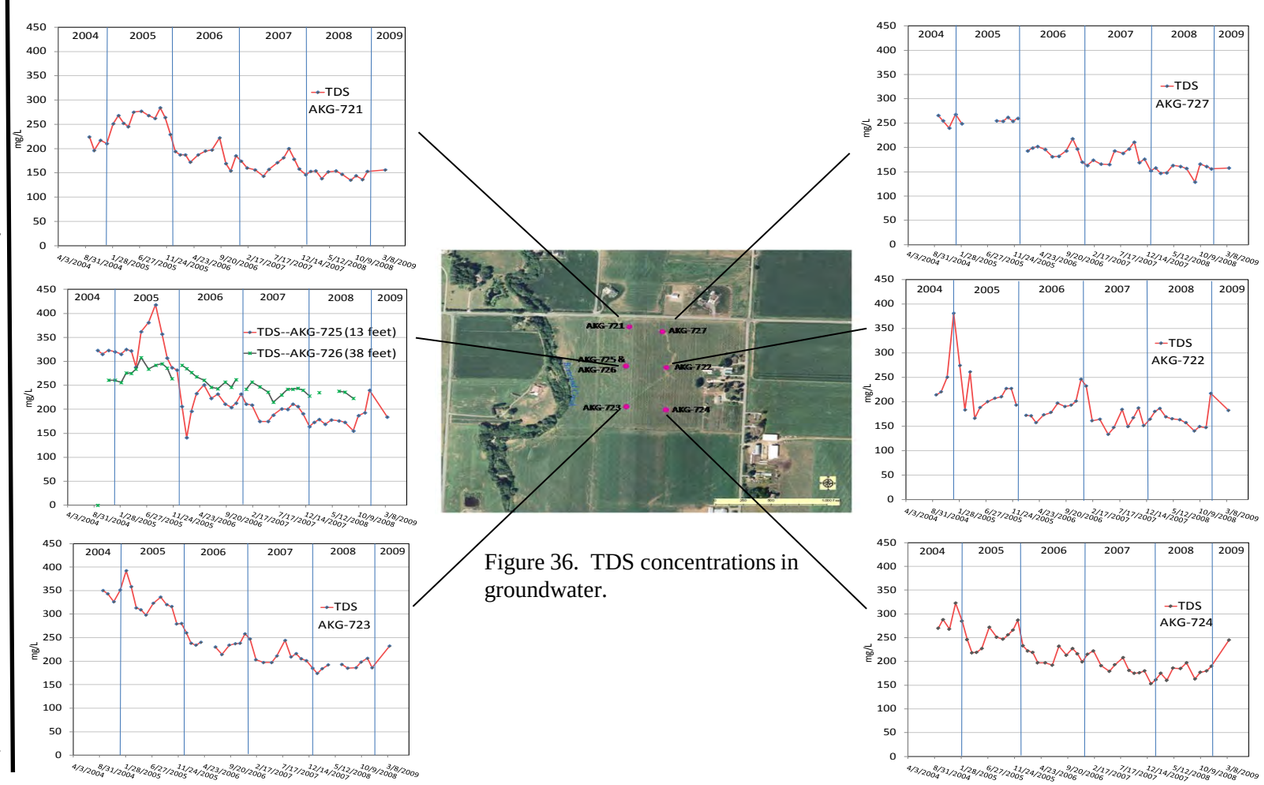


Figure 36. TDS concentrations in groundwater.

Plate 4. pH, dissolved oxygen (DO), specific conductance, and total dissolved solids (TDS) concentrations in groundwater. Concentrations are in mg/L except specific conductance in umhos/cm.

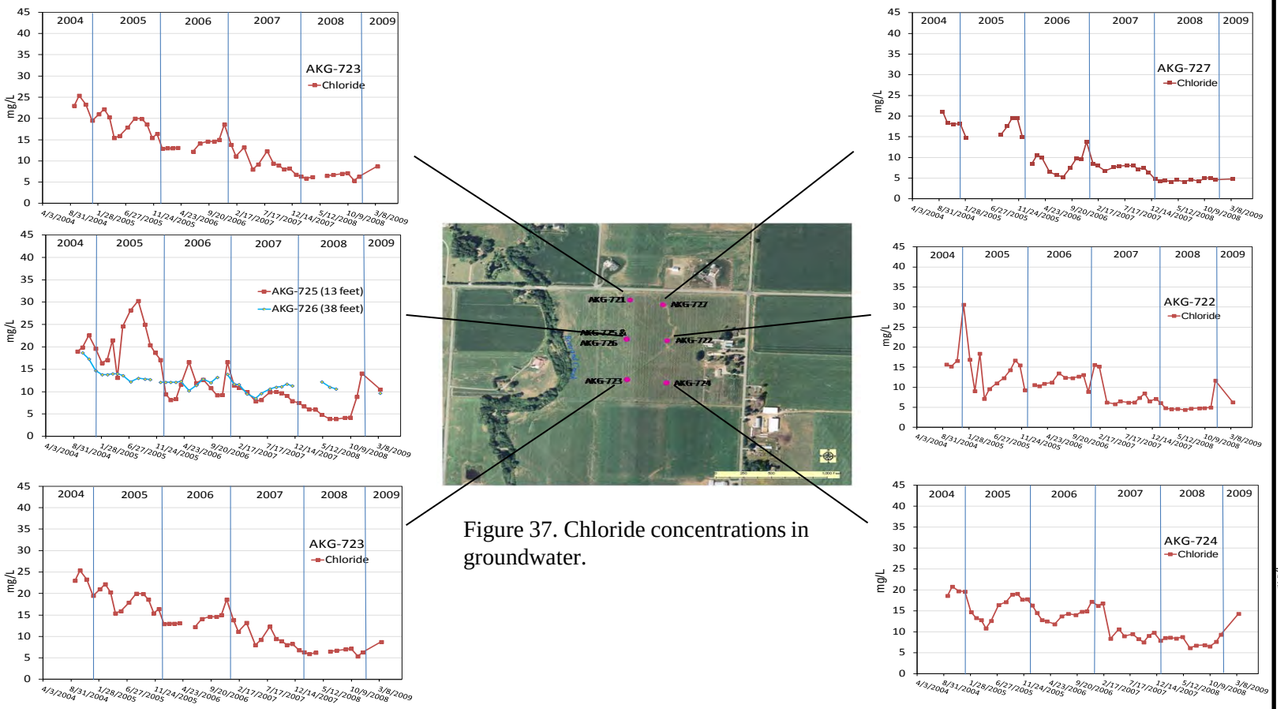


Figure 37. Chloride concentrations in groundwater.

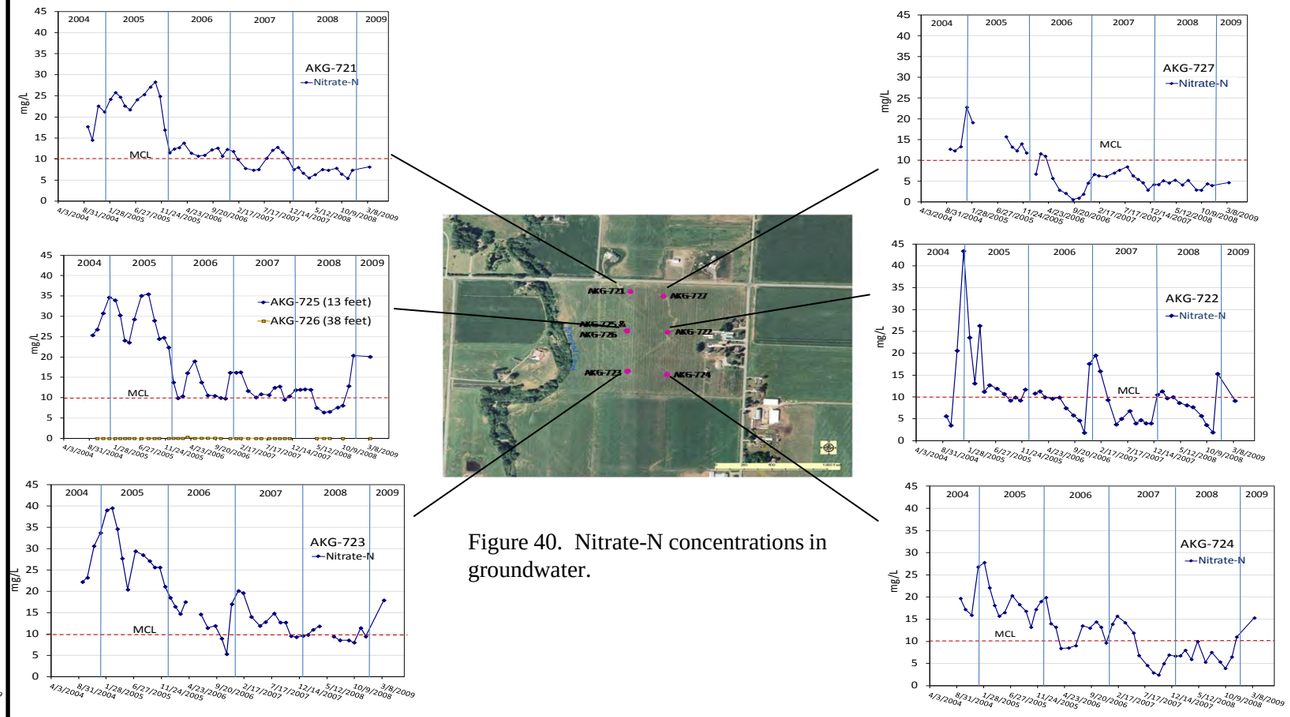


Figure 40. Nitrate-N concentrations in groundwater.

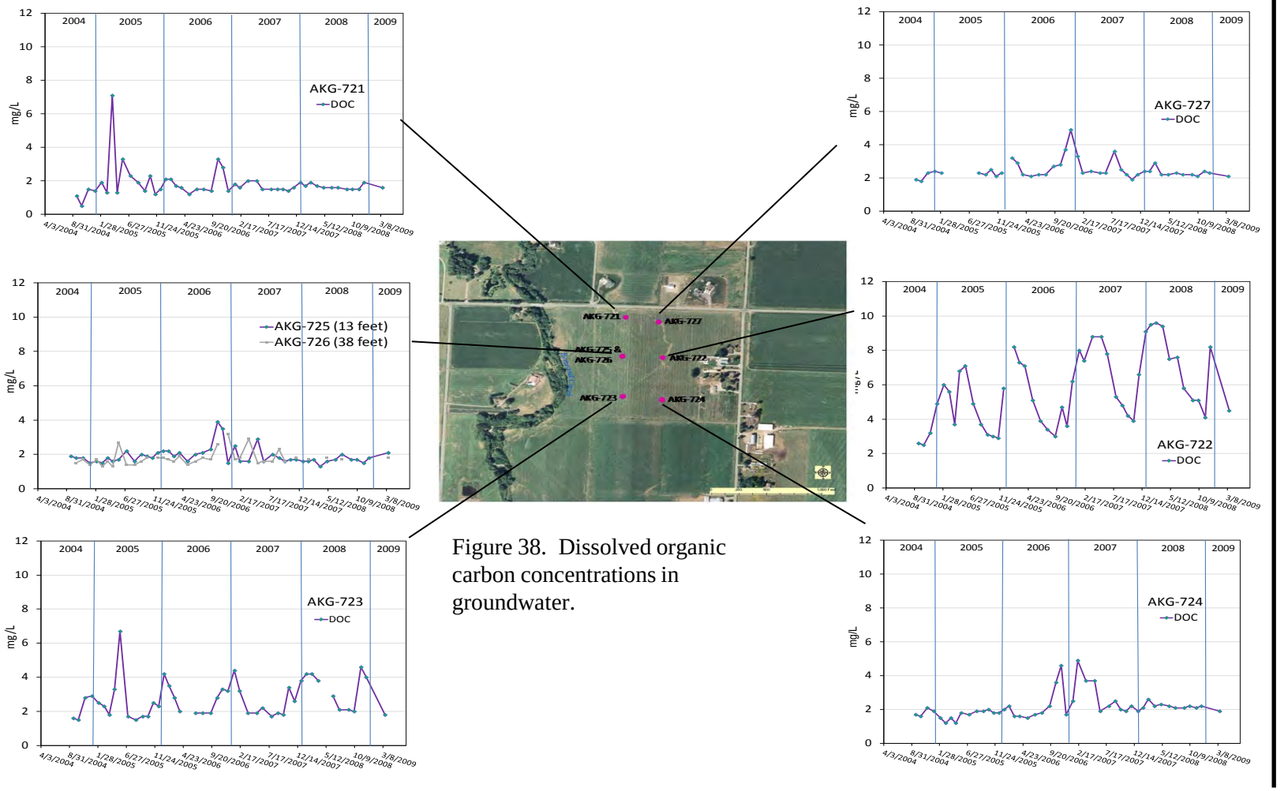


Figure 38. Dissolved organic carbon concentrations in groundwater.

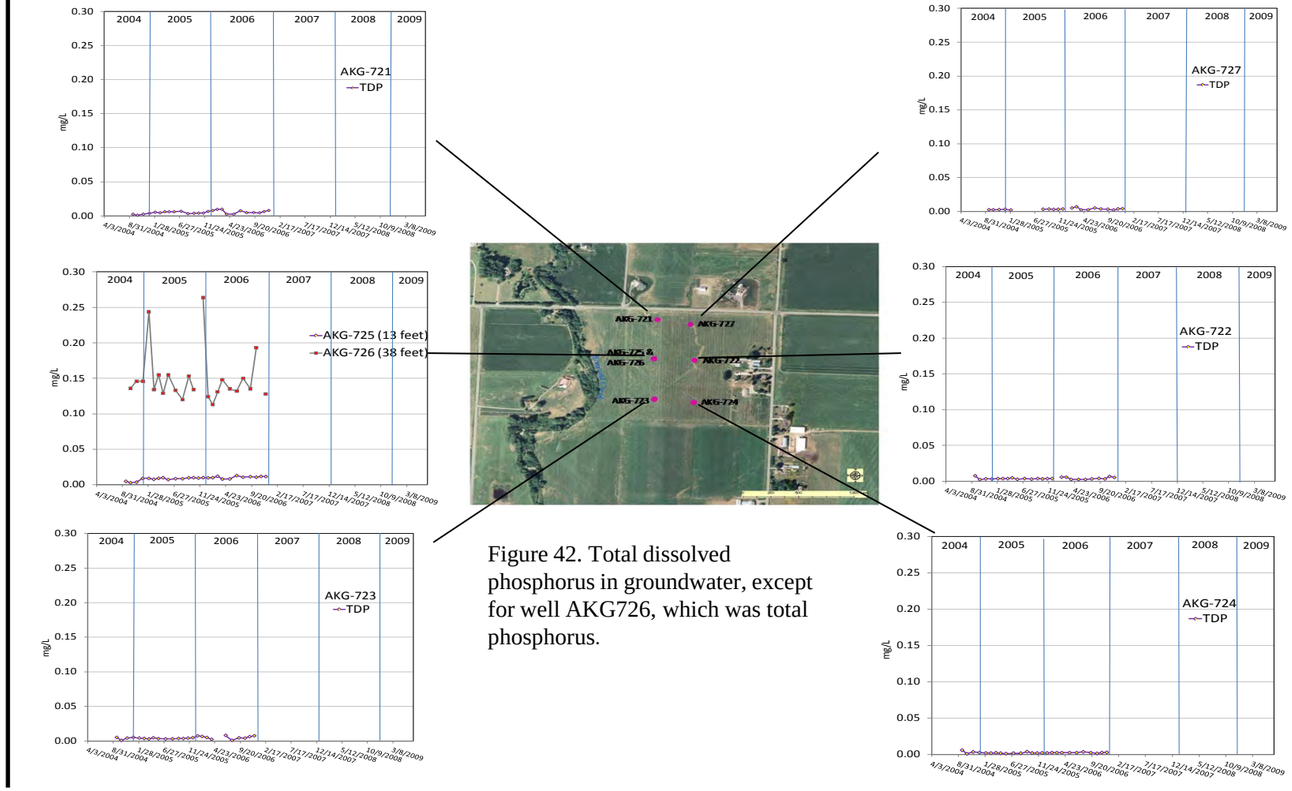


Figure 42. Total dissolved phosphorus in groundwater, except for well AKG726, which was total phosphorus.

Plate 5. Chloride, dissolved organic carbon, nitrate-N, and total dissolved phosphorus concentrations (mg/L) in groundwater.

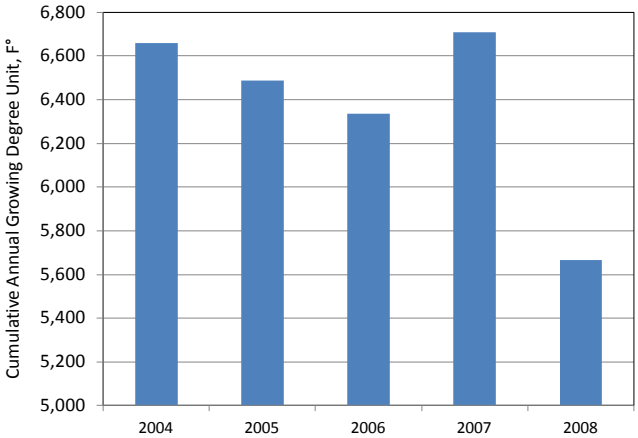


Figure 45. Annual total growing degree days from VanWieringen and Harrison (2009).

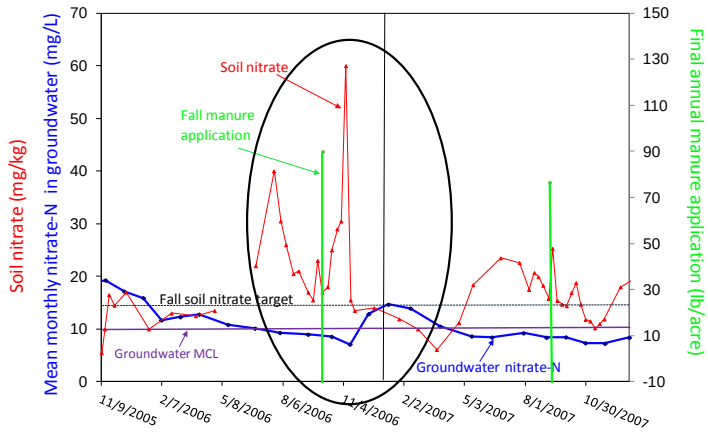


Figure 51. Soil nitrate and mean shallow groundwater nitrate-N concentrations. The circle points out the results of manure application on October 5, 2006.

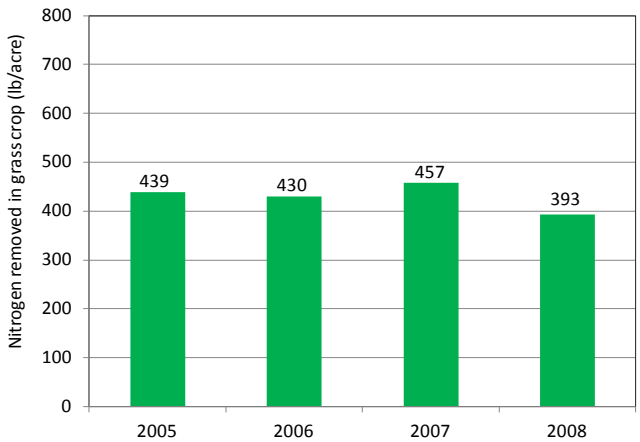


Figure 46. Annual nitrogen mass removed in the grass crop.

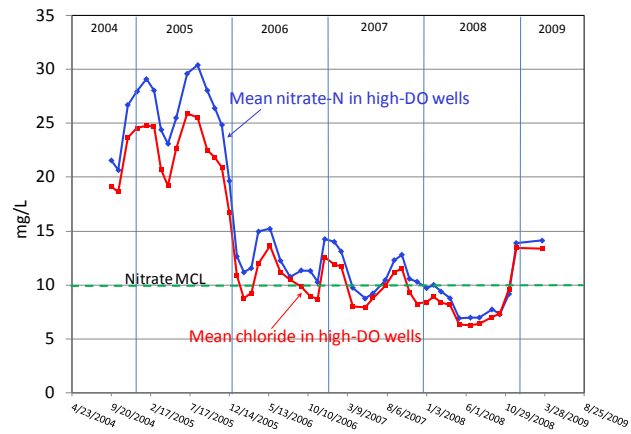


Figure 52. Nitrate-N and chloride concentrations in wells with high dissolved oxygen (DO).

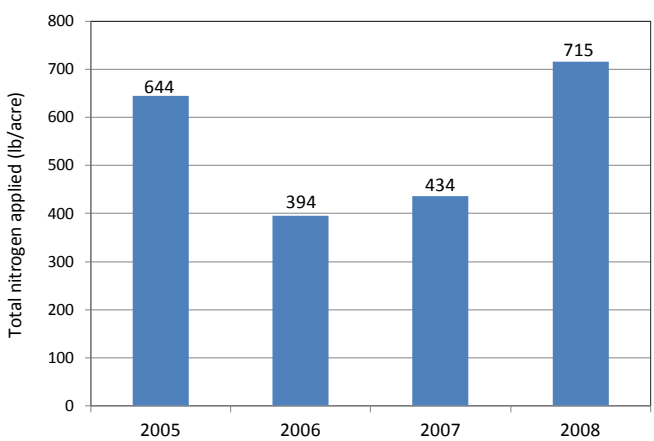


Figure 47. Total nitrogen applied annually (manure, fertilizer, and irrigation water).

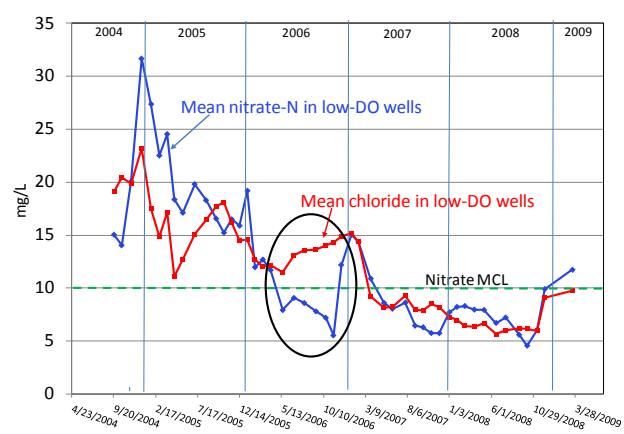
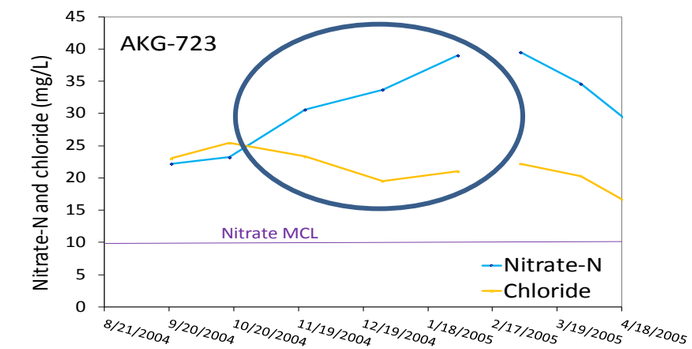
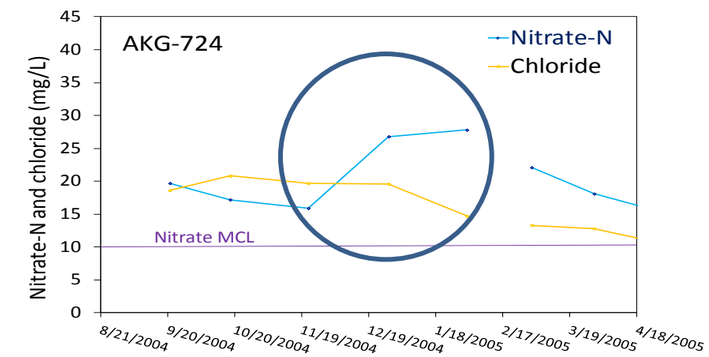


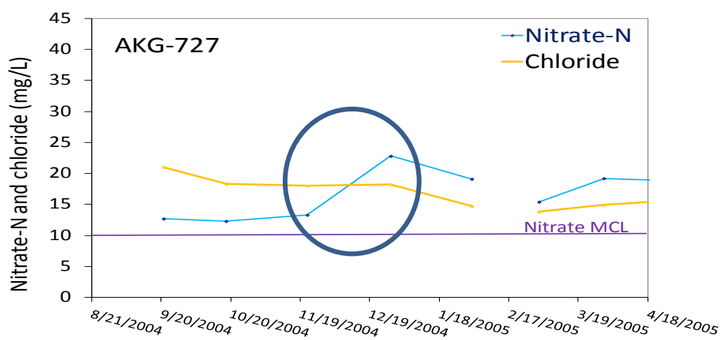
Figure 53. Nitrate-N and chloride concentrations in wells with low DO. The circle indicates a period when denitrification was probably a strong influence, because DO was less than 2 mg/L, and the water table below the root zone.

Plate 6. Discussion figures.

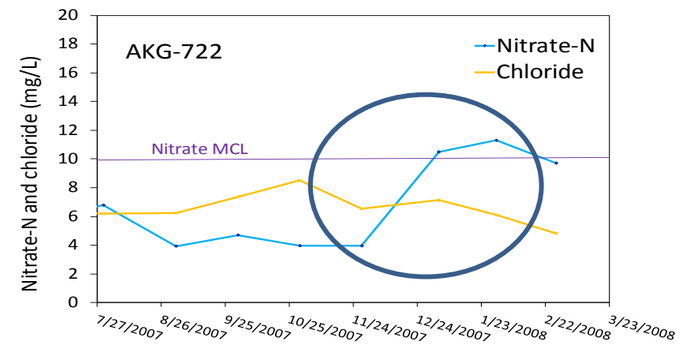
Winter 2004 to 2005

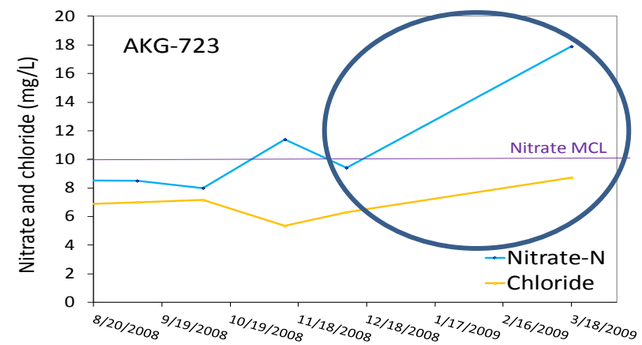


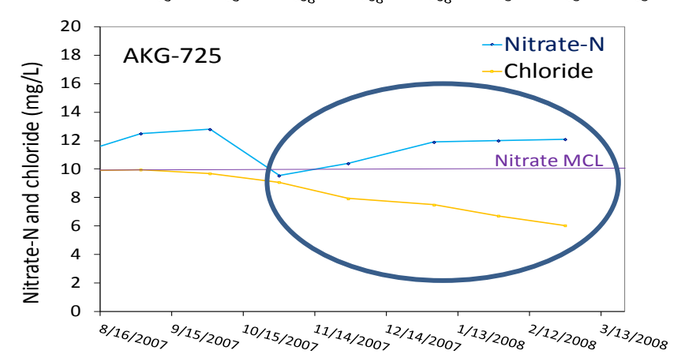




Winter 2007 to 2008







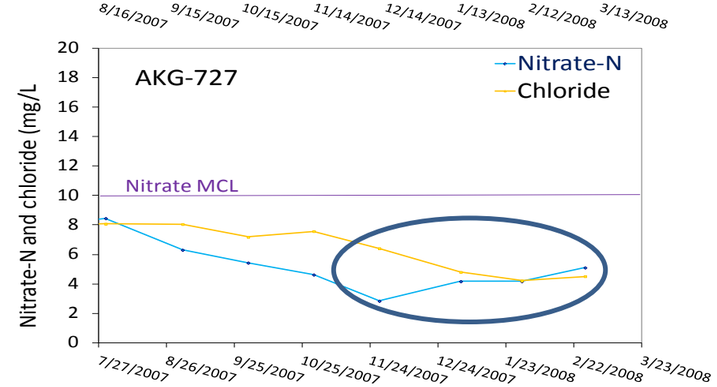


Plate 7. Nitrate and chloride concentrations during the winters of 2004-2005 and 2007-2008 in shallow groundwater. Circles indicate times in the winter when nitrate increases and chloride does not.

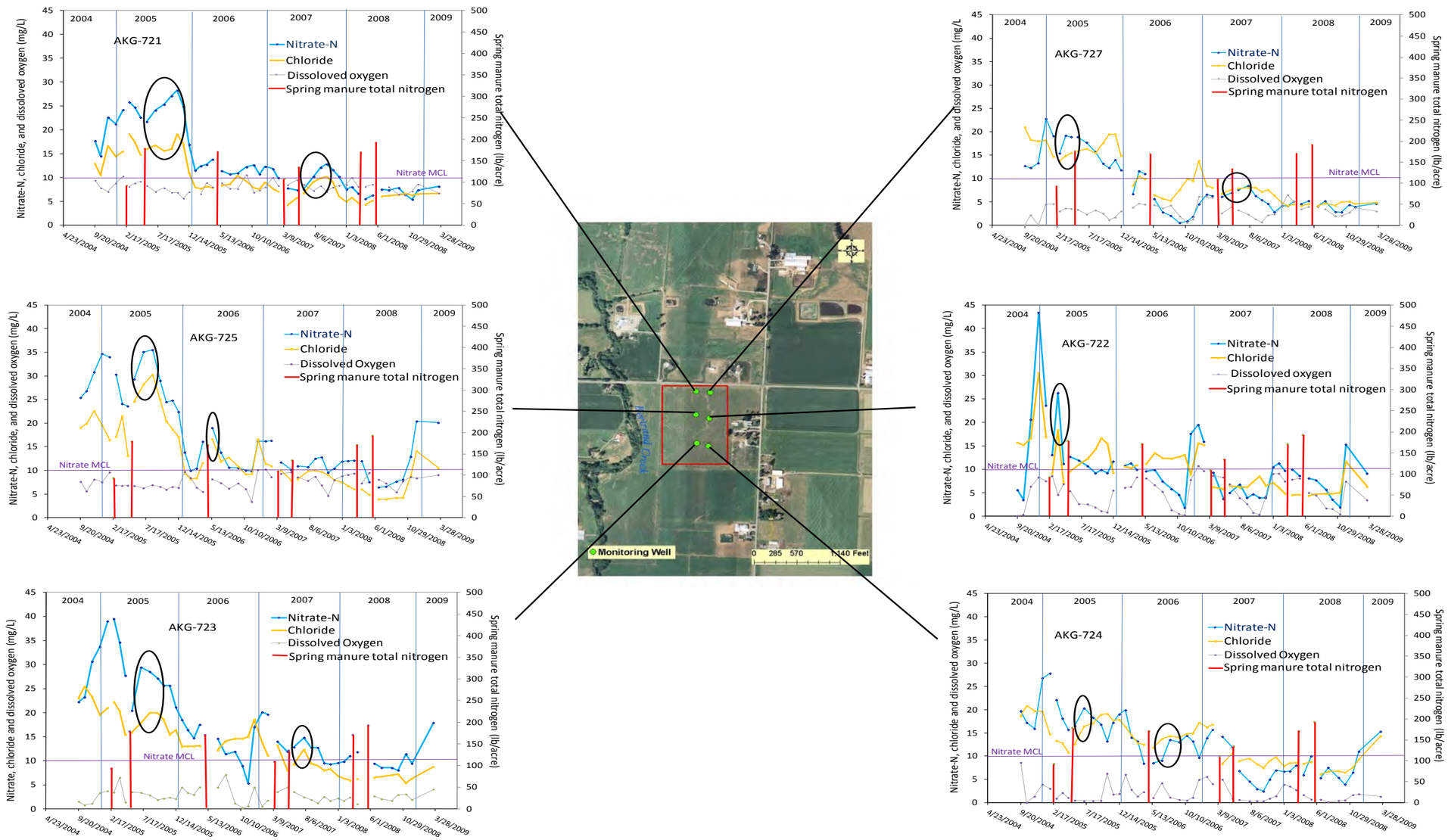


Plate 8. Nitrate-N, chloride and dissolved oxygen concentrations in individual monitoring wells in mg/L and spring manure total nitrogen applied in lb/acre. Circles show times when spring manure application was followed by elevated groundwater nitrate concentrations.

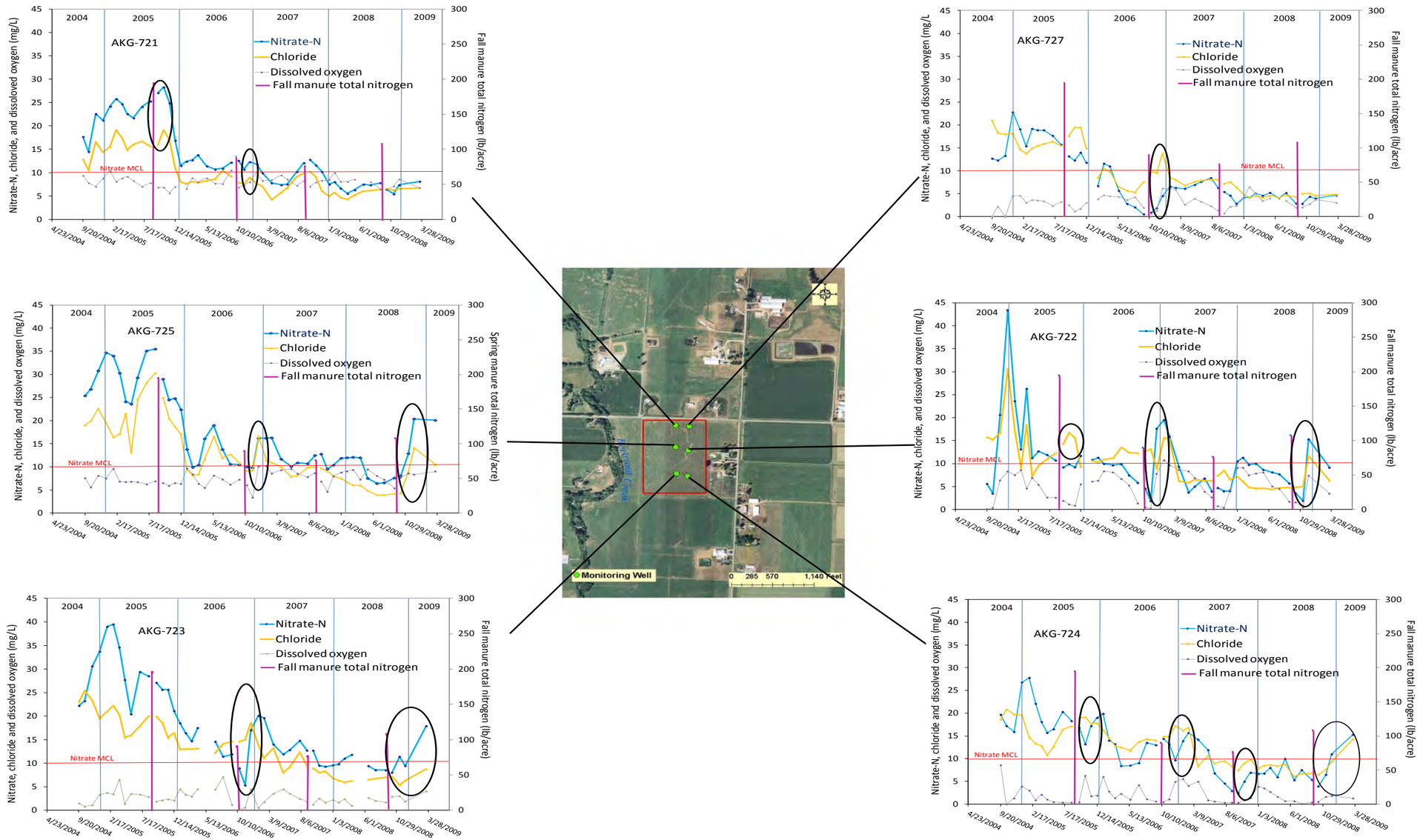


Plate 9. Nitrate, chloride and DO concentrations and total nitrogen application in fall 2005 through 2008. Circles indicate instances when chloride and nitrate increase following fall manure applications.

Appendices B through T are linked as a separate pdf on the web.

Response to Comments is also linked as a separate pdf on the web.

From:	Prest, Virginia (AGR)
To:	Fisher, Ralph; Jim Trull
Subject:	RE: GWAC - IAWG
Date:	Wednesday, May 20, 2015 1:37:23 PM
Attachments:	image001.png

Boy no kidding

Thanks

From: Fisher, Ralph [mailto:fisher.ralph@epa.gov]
Sent: Wednesday, May 20, 2015 1:34 PM
To: Prest, Virginia (AGR); Jim Trull
Subject: RE: GWAC - IAWG

Got ya. I understand its all in the "air"

Ralph

From: Prest, Virginia (AGR) [mailto:VPrest@agr.wa.gov]
Sent: Wednesday, May 20, 2015 1:26 PM
To: Fisher, Ralph; Jim Trull
Subject: RE: GWAC - IAWG

Ralph

I am sharing with this email BUT I want to advise – we have not gotten the money yet so please do be cautious when sharing this information.

Ginny

From: Fisher, Ralph [mailto:fisher.ralph@epa.gov]
Sent: Wednesday, May 20, 2015 12:23 PM
To: Prest, Virginia (AGR)
Subject: RE: GWAC - IAWG

Sounds good. Would you mind if I (or you) share that with Jim Trull. He is asking us to revise the BMP list for the GWMA. It might be good for him to know the general direction the Dept. of Ag wants to take. It help coordinate their activities with theirs.

Ralph

From: Prest, Virginia (AGR) [mailto:VPrest@agr.wa.gov]
Sent: Wednesday, May 20, 2015 12:57 PM
To: Fisher, Ralph
Subject: RE: GWAC - IAWG

Ralph

Below is my official answer I have been handing out. Lots of thoughts about how and what I should be doing. Course we have ideas as well and that is what help me put together the budget and proposal.

> I can give you an idea of DNMP's plans are if we successfully receive the two year funding for additional activities in counties that are impacted by poor water quality. The funding will be used to fund
> : additional inspector for two years in the Puget Sound Region
> : agronomic expertise in Eastern WA Region
> And training for farmers in subjects like
> : nutrient management training specific to right amount at right time in the right place (weather forecasts, recordkeeping)
> : irrigation water management and
> : other related subjects like feed management, on-farm composting, and others
> : and based on most recent court case, how to operate and maintain a lagoon with poly liner

We will be contracting the training for individual segments with conservations districts, land grant, and other technical service providers but I plan to have several little pots contracted out versus one big pot.

I hope this helps. If you have questions, give me a call.

Virginia "Ginny" Prest, Program Manager
Dairy Nutrient Management Program
Washington State Department of Agriculture
Office (360) 902-2894
Cell (360) 529-7422
vprest@agr.wa.gov
http://agr.wa.gov/FoodAnimal/Livestock-Nutrient/

From: Fisher, Ralph [mailto:fisher.ralph@epa.gov]
Sent: Wednesday, May 20, 2015 11:05 AM
To: Prest, Virginia (AGR); Peters, R Troy; Trull, Jim; (lc@sycd.us)
Subject: RE: GWAC - IAWG

So Ginny, if approved, how will funds be made available? Are they intended for cost sharing of individual practices or for educational meetings and training sessions? If they are available for individual practices do the practices have to be those listed in the NRCS FOTG, or can "component" practices be cost shared that do not fit within an existing NRCS practice?

Ralph

From: Prest, Virginia (AGR) [mailto:VPrest@agr.wa.gov]
Sent: Friday, May 08, 2015 11:19 AM

To: Peters, R Troy; Trull, Jim; (lc@sycd.us); Fisher, Ralph
Subject: RE: GWAC - IAWG

Cross your fingers and toes
If the proviso makes it through the session
We may have some finds to help.
Sent from my Windows Phone

From: Peters, R Troy
Sent: 5/8/2015 10:15 AM
To: Trull, Jim; (lc@sycd.us); Prest, Virginia (AGR); Ralph Fisher (fisher.ralph@epamail.epa.gov)
Subject: RE: GWAC - IAWG

Here are some of my ideas.

Troy

****

Cost Share for Irrigation Scheduling

Cost Share with Farmers for Irrigation Scheduling. This is a large money consumer. What level of cost share we should offer is going to be hotly debated. However as a starting point, most consultants charge from $12-$15/acre for doing irrigation scheduling (?). This runs from $1,000-$2,000 per field or site. The cost share would probably be something less than this. We should also come up with a protocol for what is required for pay-out. Ideally this would be some sort of evidence that the grower viewed and responded to the information throughout the season.

Training

One other thing that could be done that would provide a large benefit in return for low costs is for the GWAC to sponsor training events for irrigators. Growers could be trained in nutrient and water management. An estimated costs of these would be $3,000/event (?) mostly just to buy lunch for the attendees to encourage them to attend. They could be put on in conjunction with other grower meetings and the training could be done by the Benton County CD and/or WSU Agricultural Extension.

Make Soil Moisture Sensors Less Expensive

Soil moisture sensors could be provided to growers to use. This should be done on a cost-share basis instead of simply gifting the sensors so that the grower values the sensors and the information that they provide. Training is very important for the grower to know how to properly install the sensors, collect the data, and how to interpret the data to make good management decisions. Simple sensors can be purchased for about $250 for three depths and a reader, up to $2,000 for more sophisticated systems with telemetry and automatic reporting online.

ET-Based Irrigation Scheduling on Your Phone

There is a free irrigation scheduling app that runs on mobile phones http://weather.wsu.edu/ism. It automatically pulls weather data from the selected weather station and does ET-based irrigation scheduling. It synthesizes all of this information to tell the grower how much water to apply, or how long to run their irrigation system. It is freely available and in their pockets. ET-based irrigation scheduling has been shown to provide very similar benefits as soil moisture-based irrigation scheduling and is often more accurate. Additional education and outreach to show growers how to use this effectively needs to be done. Although this is a good idea and is headed in the right direction, additional work needs to be done to make it more intuitive and user friendly based on interviews with growers who are trying to use it.

Mobile Irrigation Lab

This consists of an employee that does irrigation system audits or evaluations throughout the valley. They will evaluate irrigation systems for efficiency and/or uniformity and give the grower

a report of their system operation and make suggestions for improvement. They will also provide training on nutrient management and irrigation scheduling options and techniques that are tailored to that operation or individual. This might be done on a cost-share basis for a small fee (to make it more valuable to the grower). It would be staffed by one trained individual who knows how to do the evaluations, and one temporary helper that would serve as an extra set of hands. They would need a pickup truck that contains the necessary hardware to do the evaluations. It would cost an estimated $35,000/year for the trained person, and $18,000/year for the helper. This assumes they work 8 months/year. The vehicle and fuel costs would be an estimated $6,500/year, and goods and services would be about $3,000/year. This comes to a total of an estimated $62,500/year.

From: Trull, Jim [mailto:trullj@svid.org]
Sent: Wednesday, May 06, 2015 11:34 AM
To: Peters, R Troy; (lc@sycd.us); Ginny Prest (VPrest@agr.wa.gov); Ralph Fisher (fisher.ralph@epamail.epa.gov)
Subject: GWAC - IAWG

Hi:

At various times I have had individual conversations with each of you on Irrigation Water Management and would like to develop some collective thoughts on a possible way to proceed. We are frustrated by wanting to solve problems while currently being tasked with developing a plan to solve problems. I am hopeful we can find a way to do both.

Last fall, Troy proposed a mobile lab which didn't get funded because it was viewed as implementation rather than fact finding. At the last meeting and following Ralph suggested the need to be gathering data on Irrigation Water Management parallel with our efforts on Nutrient Management. Ginny mentioned to me a possible source of finding for Irrigation Water Management which we should explore.

I want to have a Workgroup Meeting to work on BMPs (See earlier email of that issue) and on Irrigation Water Management. It won't be productive to start at ground zero at a meeting. I would like to have some basic proposals which we can refine at the meeting. I plan to schedule a meeting for some time in the next two to three weeks so could your send me your thoughts about BMPs and IWM by Monday May 18th?

I hate to burden you with extra work but you are the experts – if anyone will figure it out, it will have to be you folks!

Thanks! Jim

11 Executive Summary: Part II

Considerable effort is currently being devoted to the conservation and restoration of salmonids in the Pacific Northwest. Aquatic conservation strategies have been developed for all Federal lands in the region; however, no comparable strategies exist for protecting and restoring salmonid habitats on private lands. Nonfederal lands constitutes more than 50% of the total land area in the region, and many of the most historically productive streams and rivers flow through private lands; thus, these lands have a critical role to play in the recovery of salmonids.

Part II of this document presents an ecosystem-oriented approach to the planning and monitoring of salmonid habitat conservation efforts on nonfederal lands in the Pacific Northwest. We focus on the effects of land- and water-use practices on salmonids and their habitats and on how these impacts can be minimized through improved planning and management, but we recognize that other human activities significantly influence salmonid populations and must be addressed if salmonids are to persist over the long term. Thus, the recommendations contained herein should be considered as one part of a larger, comprehensive strategy to restore salmonids.

This document provides a conceptual framework for organizing a regional conservation strategy, guidelines for monitoring habitat conservation plans (HCPs) and other conservation efforts, and criteria by which the Agencies can evaluate habitat conservation activities. Recommendations made in this document are intended as guidelines for conservation planning, not formal requirements. Each planning situation is likely to be unique, and not all planning elements may be warranted in each case.

We propose a hierarchical approach to the development and evaluation of HCPs and other conservation efforts, stressing the need for site- or watershed-level conservation efforts to be developed and evaluated within the larger context of basin and regional conservation goals. We outline critical issues that should be addressed during HCP planning at the scales of region and basin, watersheds, and individual sites as well as specific elements that should be contained in HCPs and criteria for evaluating the potential effectiveness of HCP provisions where such criteria are supported by current scientific information. Strategies are proposed for monitoring the compliance and the effectiveness of conservation plans at levels ranging from specific sites to regions. Finally, issues related to the implementation of this conservation strategy are discussed.

11.1 Ecological Goals of Salmonid Conservation

A restoration strategy to ensure the long-term persistence of salmonids will be most effective if it is grounded in principles of watershed dynamics, ecosystem function, and conservation biology (reviewed in Part I). We believe that five fundamental ecological goals should underlie salmonid conservation activities at all levels, from site-specific management prescriptions to watershed plans to regional recovery efforts. These goals include

- Maintain and restore natural watershed processes that create habitat characteristics favorable to salmonids.
- Maintain habitats required by salmonids during all life stages—from embryos and alevins through adults—and maintain functional corridors linking these habitats.
- Maintain a well-dispersed network of high-quality refugia to serve as centers of population expansion.
- Maintain connectivity between high-quality habitats to allow for reinvasion and population expansion as degraded systems recover.
- Maintain genetic diversity and integrity within and among salmonid stocks and species.

Activities that maintain and restore natural watershed and ecological processes, facilitate the expansion of refugia, enhance connectivity between refugia or from headwaters to the ocean, and allow full expression of the genetic potential of the species should be encouraged; those activities that do otherwise should be discouraged.

11.2 Planning Elements

Ecosystem-oriented approaches to land and resource management are being recommended by scientists and the management agencies that oversee activities on public and private lands. Although the term "ecosystem management" has been defined a number of different ways in the literature, the goal of preserving ecosystem integrity while deriving sustained benefits for human populations is common to most definitions. For Habitat Conservation Plans or other conservation agreements to succeed, it is important that they be developed and evaluated within the context of larger ecosystem restoration strategies. In this regard, a broad spectrum of issues should be addressed: site-specific impacts: cumulative effects of multiple activities (in space and time) throughout a watershed; the distribution and status of salmonid species or population segments at region, basin, watershed, and site levels; and the status of other biota and resource values.

Implementation of ecosysteni-oriented approaches to land management requires a hierarchy of planning scales, including regions, basins or provinces, watersheds, and individual sites. Watershed analyses and site prescriptions that are the most likely components of conservation plans should be imbedded within analyses at larger spatial scales. This is critical for salmonid conservation efforts because 1) the distribution and environmental requirements of salmonids typically extend beyond watershed boundaries and 2) the spatial context within which a particular watershed lies is an important factor for evaluating the potential ecological effects of land management activities. In the remainder of this section, we identify what we believe to be key planning elements at various levels in the planning hierarchy that should be involved in the preparation and evaluation of HCPs or other conservation efforts.

11.2.1 Region and Basin (or Province) Levels

A number of important issues and goals transcend watershed boundaries and thus cannot be satisfactorily addressed without basin-level and regional assessments. These include protecting aquatic biodiversity or ecosystems with unique physical or biological attributes; identifying and protecting threatened, endangered, or other at-risk species or stocks that may be adversely affected by the proposed activities; determining the role of the affected stream or watershed in fostering connectivity between existing refugia (e.g., Federal key watersheds, salmon core or source areas, aquatic diversity areas); maintaining proper function of migration corridors used by anadromous salmonids and enabling dispersion of resident species; assessing the current and historic potential of the affected area to produce salmonids and the potential for restoration if degraded; identifying the primary natural and anthropogenic stressors presently occurring and projected to occur within the basin and determining if these will be exacerbated by the proposed activities: and assessing the potential for the proposed activities to affect estuarine environments into which anadromous salmonids enter.

Three initial steps by the managing Agencies would facilitate attainment of these broader goals: 1) establishing a network of key watersheds on private lands to complement Federal key watersheds; 2) adopting riparian protection standards for all riparian areas across the four-State area: and **3)** delineating evolutionarily significant units (ESUs) for all species of anadromous and resident salmonids. The establishment of key watersheds is needed to ensure that regional conservation goals are not adversely affected by site- and watershed-level decisions on nonfederal lands. Protection of the riparian zone is essential for maintaining many stream processes, moderating the influence of upland management on aquatic systems, re-establishing connectivity between fragmented habitats and biotic refugia, and maintaining ecologically functional migration corridors linking headwater streams to the ocean. The delineation of ESUs is needed to clarify biodiversity objectives, which in turn should be considered when designating key watersheds.

11.2.2 Watershed Level

Watersheds with areas of approximately 20–200 square miles are generally the most practical for planning and analysis: it is at this level that linkages between physical and biological processes can be addressed most effectively. Watershed analysis serves several important functions: 1) it offers a means of addressing cumulative effects of multiple activities within a watershed on ecological processes and aquatic habitats; 2) it provides an assessment of current conditions within the watershed, which allows existing resource problems to be identified and future activities to be planned in a more ecologically sound manner; 3) it helps to identify specific portions of the watershed highly sensitive to human disturbances and allows prescriptions to be developed appropriately for the level of sensitivity; and 4) it provides information that helps to refine our understanding of physical and biological processes and how these vary across the landscape—information that can then be used to develop ecoregion- or basin-level standards or criteria. Watershed analysis can also help identify and prioritize habitat-restoration opportunities.

We recommend that watershed analysis be a key component of conservation planning on nonfederal

lands. Conservation plans should, at a minimum, incorporate evaluations of how proposed activities will potentially affect hydrology (total water yield, peak flow, base flow, and seasonal timing), sediment transport (mass wasting and surface erosion), riparian functions (LWD recruitment, small organic litter inputs, stream shading, bank stabilization, and nutrient cycling), channel condition (bed morphology, substrate type, and physical structure), and water quality (temperatures, dissolved oxygen, and pollutants). Watershed-level analyses should also be conducted to assess biological conditions in the watershed, including fish distributions, status, and habitat conditions. Although specific resource issues are discussed individually, conservation plans should strive to integrate various analyses because of the strong linkages among processes.

Specifically, we recommend HCPs and other conservation agreements should contain the elements listed below.

- **Conservation plans should iiiclude a strategy for preventing cumulative hydrologic effects within the watershed or watersheds.** Land- and water-uses can substantially influence the amount and timing of water delivered to the stream channel, resulting in changes in total water yield, peak flows, summer base flows; and seasonal timing of flows. Conservation plans should specifically address each of these issues by minimizing the areal extent of vegetation disturbance, the area in hydrologically "immature" condition, and the areal extent of roads and other impervious surfaces. Provisions should be included for protecting summer-low flows and seasonal flushing flows, and for reducing irrigation withdrawals where inadequate flows are of concern.

- **Conservation plans should iiiclude a long-term plan for minimizing cumulative sediment delivery to streams.** Land-use activities substantially alter the rate at which sediment is delivered to streams via both mass wasting and surface-erosion processes. Conservation plans should contain provisions to minimize or avoid land-use activities in areas susceptible to mass wasting and surface erosion as well as in riparian zones; minimize total road density within the watershed, including limited entry to roadless areas; develop a road maintenance schedule to prevent and mitigate effects of sediment; and actively rehabilitate roads no longer in use, particularly those in riparian areas. Plans for minimizing impacts of sediment should be based on a thorough assessment of existing and potential erosion and mass wasting problems within the watershed, with the goal of identify areas within the watershed that are at high risk for erosion.

- **Conservation plans should include a comprehensive strategy for protecting riparian areas along all streams.** Riparian vegetation provides shade and moderates stream temperatures, contributes Iarge woody debris to streams, adds small organic matter to streams, stabilizes streambanks, controls sediment inputs from surface erosion, and regulates nutrient and pollutant inputs to streanis. Removal of riparian vegetation diminishes each of these critical functions. All HCPs should establish riparian buffers designed to maintain the full array of ecological processes needed to create and maintain favorable conditions through time.

- **Conservation plans should include a comprehensive strategy for maintaining water quality.** High water quality is required by salmonids during all life stages and can be degraded by land-use and water-use activities. The goal of the water-quality plan should be to maintain temperature, dissolved, nutrients, and other dissolved materials (including toxic substances, where applicable) within the natural range of variability for the particular body of water and time of year. A thorough assessment of current conditions within the watershed is needed to develop this strategy. This assessment should seek to identify acute water quality problems within the watershed, identify specific factors that contribute to these problems, and compare current temperature regimes with reference conditions.

- **Conservation plans should contain a watershed-level strategy for minimizing the impact of roads on aquatic ecosystems.** Roads frequently are the dominant liuman-caused source of sediments delivered to streams, and they influence the routing of water from uplands to the stream channel. In addition, when placed near streams, roads often simplify channels, alter hydraulic processes, and prevent natural channel adjustments. The road strategy should include the development of a long-term transportation plan, regularly scheduled maintenance, replacement of inadequate road culverts, and removal and rehabilitation of roads that are unneeded or that degrade salmonid habitats.

- **Conservation plans should include an assessment of salmonid distributions and status.** The ultimate goal of habitat conservation plans is to protect habitats required by salmonids during all life stages. Identifying important

salmonid habitats is critical to the development of specific management strategies and prescriptions. Goals of this assessment should be to 1) identify all habitats accessible (existing or potential) to salmonids, 2) document the distribution and abundance of wild salmonids by species and life stage, 3) identify areas of high productivity or importance for specific life stages, 4) determine trends in salmonid abundance within the watershed, and 5) document past and present indtroductions from hatcheries to waters within the watershed,

- **Conservation plans should include an assessment of current channel conditions and physical habitat.** Channel conditions and physical habitats of salmonids have been altered directly through channelization, revetments, stream cleaning, and dam construction, and indirectly through changes in hydrology, sediment loading, and recruitment of large woody debris. The goals of the habitat assessment should be to characterize channel forms and geomorphic processes affecting channels in the watershed; to identify reaches that are sensitive to large variation in runoff, sediment supply, and large woody debris; to identify reaches that have been subject to human-caused and natural disturbances; and to evaluate the effects these disturbances have had on sensitive reaches and to assess the degree of recovery.

11.2.3 Site Level

The landscape- and watershed-level analyses proposed in this document provide the context from which site-level prescriptions can be made that will effectively protect salmonids. Knowledge of existing watershed conditions and resource problems, as well as the potential sensitivity of different areas of the basin or watershed to land use activities, will enable owners of nonfederal lands to avoid undesirable effects on salmonids and their ecosystems.

Conservation plans should incorporate site-specific prescriptions that accurately reflect the resource concerns identified for the affected area. Uniform prescriptions are generally inappropriate; nevertheless, certain practices are inherently less disruptive to ecosystems than others and should be employed to the degree possible. These best-management practices are discussed by land-use type in the main body of the document, but for brevity are presented here under categories of land alteration, roads, riparian buffers, channel modifications, water use, and water quality.

Land Alteration

Emphasis should be given to minimizing the areal extent and intensity of disturbance to vegetation and soils. Logging-rotation schedules, grazing, farming, mining, and urbanization should be adjusted to minimize the total area in a disturbed state at any given time to minimize cumulative hydrologic effects. Logging should be avoided on areas identified as high risk for mass failures, such as those with steep (> 30°) or unstable slopes. Similarly ranching, agriculture, urbanization, and mining should be precluded from erosive and floodprone areas. Selective tree harvest is recommended for areas identified as moderately sensitive, while ground-based logging equipment is advised only in low-risk areas. We recommend against the burning of logging slash, favoring its retention to control surface erosion except where it increases fire risk. Where range conditions are not good-to-excellent, we recommend suspension of grazing until vegetation has recovered. Once conditions have improved, grazing strategies should be adjusted to preclude deterioration. Where surface erosion is evident, mulching is recommended until vegetative cover is restored. Areas identified as highly erosive should be retired from agriculture. Mining lands denuded of vegetation should be revegetated quickly to reduce erosion. Where chemical constituents of mine spoils inhibit recovery, spoils should be treated to ensure successful re-establishment of vegetation. The most effective means for minimizing urban impacts is through strict State, county, and city land-use planning. Construction should be avoided on steep hillslopes and seasonal wetlands.

Roads

Regardless of land-use type, we reconmend placing roads away from streams, riparian areas, or wetlands; avoiding unstable hillslopes or areas where risk of sediment delivery to streams is high; avoiding stream crossings; installing culverts adequate to allow year-round passage of fish; reseeding and stabilizing areas disturbed during construction; ensuring adequate drainage from road surfaces to minimize erosion; and regularly maintaining drainage ditches and culverts. We also encourage obliteration and revegetation of problem roads and removal or replacement of inadequate culverts. Alternative forms of urban transportation should be promoted to reduce the need for additional roads.

Riparian Buffers

Regardless of land-use type, riparian buffers are reconmended on all streams; their dimensions will depend on the setting and level of protection desired. **An** evaluation of appropriate buffer widths for protecting critical riparian functions and a review of State and Federal forest-practice rules is presented in the main text. Similar buffers are needed for

nonforest lands, and niay require planting native riparian vegetation in highly disturbed agricultural, range, mining, and urban areas. Only those activities that can be performed without adversely affecting natural riparian functions or values should be allowed in buffers. We recommend that grazing be excluded through fencing or removal of livestock in all riparian areas where function of riparian vegetation is impaired. Once recovery has occurred, riparian grazing should be limited in duration and intensity to ensure these functions are maintained. Facilities for watering livestock should be located away from the stream channel and riparian zone, where possible. Where riparian areas are fenced, small access areas for livestock may be appropriate if unlikely to degrade the stream. Conservation can be further enhanced by retiring converted wetlands from agriculture. Urban riparian areas and wetlands that have not been developed should be preserved and no new development allowed. Where feasible, impervious surfaces should be removed and vegetation restored.

Channel Modifications

Where feasible, we recommend removal of dams and rip-rap structures, as well as reintroduction of beaver. In general, we recommend against instream manipulations, such as placement and cabling of logs or other artificial structures, because of high cost per mile and high likelihood of failure or adverse consequences. These structures should only be viewed as stopgap efforts in special situations, not as mitigation for poor management practices. Reconnecting streams to off-channel areas has greater potential for restoring salmonid abundance, but natural riparian recovery through revegetation is emphasized.

Water Use

New water allocations should be approached with great caution, while increased instream water rights are needed for fisheries. All water diversions from salmonid streams should be screened to prevent entrainment. For streanis with diminished water quality or quantity, a watershed conservation strategy should be developed to reduce the volume of water needed for agriculture. Drainage structures should not be used unless combined with irrigation from deep groundwater. Water for mining purposes should not be withdrawn from streams supporting at-risk salmonids or habitats identified as critical for salmonid production. **A** conservation strategy for mining water should be developed, including treatment and recycling of wastewaters and reductions in groundwater pumping where streaniflow niay be affected. Where urban water withdrawals are degrading salmonid habitats, water conservation and recycling should be promoted.

Water Quality

Regardless of land-use type, chemical treatments should be applied only outside riparian zones (including those of headwater streams), and aerial spraying should be conducted to prevent drift into the riparian zone. Where drainage ditches and tiles exist, intensive use of fertilizers or pesticides should be avoided. Organic farming and integrated pest management should be encouraged where water quality has been degraded by agricultural chemicals. We recommend against mineral or aggregate mining in streams or riparian areas of streams containing salmonids or that drain into salmonid habitats. Mining should be avoided where tailings and wastewater have the possibility of entering aquatic systems. Wastewaters should be treated and recycled on site, and waters not clean enough for re-use should not be discharged into streams. Control structures should be used to retain toxic materials and should be built to withstand extreme precipitation and geological events. Spoils containing toxic materials should be buried below the plant-rooting zone so that these materials are not absorbed by plants or carried by ground water and subsequently released into the environment. In urban areas, stormwater should be routed through waste treatment facilities, and the use of chemical pesticides and fertilizers should be discouraged.

11.3 The Role of Monitoring in Salmonid Conservation Activities

'Monitoring of salmonid conservation activities is critical for ensuring that provisions of conservation agreements are being met (implementation monitoring), that implementation of conservation plans is having the desired effect on aquatic ecosystems (assessment monitoring), and that there is an adequate information base for modifying plans if necessary to protect salmonids and their habitats (adaptive management). In this document, we propose a monitoring strategy designed to assess the condition and detect statistical trends in aquatic ecosystems at spatial scales from site to region. Sampling designs and indicators are recommended to track trends in physical, chemical, and biological conditions in uplands as well as in riparian areas and streams so that critical planning elements are monitored at appropriate spatial and temporal scales.

Long-term monitoring of salmonid conservation activities is essential to document the decadal trends in ecosystem conditions that occur in response to natural and anthropogenic disturbances and to allow separation of the effects of human activity from

natural variation. Multiscale monitoring is important to assess the effects of management activities at the scales of the site or the reach as well as to address cumulative effects at the level of catchments, basins, ecoregions, and multi-State regions. Interdisciplinary monitoring is needed because ecosystems are complex aggregations of biotic and abiotic components. Monitoring should be inter-institutional because lands are held by many different institutions, both public and private, and because many agencies have regulatory and management missions that directly or indirectly relate to salmonid conservation.

An effective monitoring program will require a computerized database-management system conducive to data entry, storage, retrieval, analysis, and reporting. Organizing a successful monitoring program of such complexity requires considerable Federal coordination and leadership but also must involve close coordination with State, tribal, and local governments, as well as private interests; this is essential both to ensure consistency of information and to take advantage of existing programs and information resources. The Research and Monitoring Committee for the President's Forest Plan is currently examining how to implement such a program on Federal lands; extending this effort to nonfederal lands in the Pacific Northwest would greatly enhance salmonid conservation planning.

11.3.1 General Monitoring Framework

Eight activities provide a framework for monitoring salmonid conservation efforts: 1) develop a set of assessment questions or objectives that the monitoring should address; 2) determine the indicators that will be used to assess biotic and abiotic conditions as well as ensure that these indicators can be related to the ecological values, the natural and anthropogenic stressors, or both; 3) use the index concept in selecting the sampling period, sampling sites (e.g., streams) and sampling locations at the sites as well as in data analysis (i.e., focus data collection and analysis on particular times, places, and indices.); 4) develop a sampling design that is appropriate for answering assessment questions (item 1 above); 5) establish reference conditions against which conservation efforts may be measured; 6) apply the data to answer resource management questions or to develop new assessment questions; 7) evaluate the effectiveness of the strategy and its results; and 8) identify ecosystem elements and processes requiring additional research.

77.3.2 Monitoring Implementation and Effectiveness of Conservation Plans

All HCPs and other conservation agreements should include an approved and consistent implementation monitoring program, by which the Agencies can determine if landowners are complying with provisions of the conservation plan. Most HCPs prepared using this guidance will involve monitoring the implementation of land-use controls to reduce hydrological modifications, sediment transport, and riparian disturbance, and many will contain provisions to improve water quality and physical habitat structure. Indicators should be measured through remote sensing and site visits (e.g., range condition, riparian tree-retention requirements). To be most effective, baseline data should be collected before conservation activities begin, and all data should be entered into a database to facilitate tracking of progress.

All conservation plans should involve monitoring to assess the effectiveness of land-use controls in restoring and protecting salmon and salmonid habitats. The focus of the monitoring should be on the aquatic and riparian ecosystems and should include physical, chemical, and biological indicators. As with implementation monitoring, consistent design and indicators should be used to the degree possible. Both remote sensing and site visits are needed. as is a large database management system.

11.3.3 Sampling Design for Monitoring Implementation and Assessment of Conservation Plans

For monitoring habitat conservation activities, we recommend a multi-State, regional, sample survey. This survey design is recommended for several reasons. 1) There are ecoregional patterns in biotic and abiotic factors, and it takes a regional approach to assess this variability. 2) Summarizing segment-level information in an organized manner facilitates making landscape-level statements, which are important for regionally distributed organisms like salmon. 3) It will be extremely expensive to inventory or census all nonfederal lands and stream miles in the region with the quantitative indicators needed to accurately and precisely assess status and trends. 4) Regional assessments of status-and-trends should be conducted in a statistically consistent and unbiased manner. 5) Fragmentary monitoring fosters fragmentary ecosystem management and social systems. 6) Previous site- and catchment-specific assessments are a key reason that it took so long to determine the extent of deteriorating stocks.

A regional sample survey or census is also important for placing individual conservation activities into an ecoregional and basin context. Such a survey can help establish reference conditions for determining desired directions and outcomes for restoration, for setting quantitative criteria for evaluating progress, and for assessing the effectiveness of conservation plans. A regional sampling effort is also needed to determine if trends

in assessed variables result from the effects of the HCP or from changes in climate, fish passage, harvest, and hatcheries. Furthermore, a regional assessment provides a basis for determining the relative condition of various watershed and stream reaches in HCPs. Regional-scale monitoring can generate important data to establish standards for specific habitat attributes. And finally, a regional approach would help standardize sampling designs and methods among the managing agencies, allowing for greater efficiency in sampling and analysis.

We propose that the Agencies adopt something like EPA's EMAP sampling design. This design is easily intensified if detailed information is needed for a single HCP or basin, yet it offers great cost savings by not requiring intensive inventorying of entire drainages. In addition, the EMAP design facilitates accurate and precise inference about resources throughout the region of concern. Equally important, EMAP's randomized design and its monitoring frequency offer rapid assessment of regional status and trends, which would be exceedingly costly or time consuming via an inventory approach.

11.3.4 Physical, Chemical, and Biological Indicators

Quantitative indicators are needed to ensure that ecological signals are discriminated from spatial, temporal, and methodological variances, thereby aiding rapid detection of trends and accurate estimates of status. Linkages between major planning elements and the recommended indicators should facilitate adaptive management and modifications in conservation plans when results deviate from expectations.

Several indicators or indicator groups have been found to be precise and responsive to stressors, especially when data are composited and metrics are integrated into multinietric or multivariate indices. Indicators that should be monitored at all assessment monitoring sites include measures of landscape condition, physical and chemical habitat variables in streams and riparian areas, benthic macroinvertebrates, and aquatic vertebrates. Monitoring of microbial respiration is recommended for urban and mining streams, and sampling of periphyton is recommended for streams on agricultural and range lands. Monitoring of salmonid genetics, spawning, and rearing should be conducted in random subsets of streams. Multiple indicators should be sampled at as many sites as possible.

11.3.5 Other Monitoring Issues

An important goal of a regional monitoring program is to identify and protect streams and catchments that are in very good condition or highly productive of salmonids. These areas are important as reference sites, biological refugia, sources of high quality water, or locations for studying natural rates of ecological processes. In addition, information obtained from reference sites may prove useful in refining criteria or standards to more accurately reflect variability across the landscape.

Several programmatic concerns should be incorporated into an effective monitoring program. Although not all of the indicators discussed above need to be monitored at all sites, it is critical that indicators and monitoring protocols be consistent among conservation plans to allow integration and analysis at broader spatial scales. To this end, all monitoring personnel should receive consistent training, and repeat sampling should be conducted at a subset of locations by other persons to ensure among-watershed comparability and to assess sampling variance. To evaluate ecoregional and basin patterns, watershed-scale data must be aggregated to the larger spatial scales; this will require coordination by the Agencies. Finally, procedures will be needed for disseminating the results of monitoring to other agencies and to the public.

Additional monitoring and assessment are desirable to attain salmonid conservation goals. These include 1) consistent probability-based survey designs and sampling methods (across all States) to more accurately estimate salmon spawning or escapement; 2) monitoring of smolt production at randomly located traps; 3) rigorous stock assessment (through genetic and morphometric analyses) of salmonids in all sub-basins of the Pacific Northwest to aid in delineating ESUs and to address biodiversity issues: 4) assessment of the influence of salmonid diseases within basins (also important in defining ESUs); 5) delineation of aquatic diversity areas and key spawning areas throughout the Pacific Northwest region to help prioritize restoration efforts; 6) continued monitoring of adults and smolts at dams and hatcheries to track trends in abundance; 7) continued monitoring of salmon harvest to document its effects on salmonid populations; 8) development of a central fish database of historical information. Monitoring of these variables also requires integration with the monitoring discussed above.

11.4 Implementation Strategy

Successful conservation and restoration of salmonid habitat in the Pacific Northwest will require that individual conservation efforts, such as HCPs, be integrated into a comprehensive regional program. An important part of such a program will be identifying who is responsible for developing habitat conservation plans, monitoring the implementation and effects of those plans, and evaluating the overall effectiveness of the program. Most of this chapter focuses on these issues. Additional issues that will

likely arise during conservation planning are briefly discussed.

11.4.1 Development of HCPs and a Regional Conservation Strategy

It is clearly the responsibility of landowners and land managers, with Agency guidance, to develop conservation plans at the site or watershed scale. For watersheds with single ownership, this process is relatively straightforward; however, where conservation efforts involve multiple ownerships or mixed private and government ownerships, the process becomes more complex. In such cases, two strategies are recommended. Where there are dominant or codominant owners, we recommend that they take the lead in HCP preparation, with contributions from fellow landowners proportionate to ownership. Where ownership patterns are more heterogeneous, watershed councils or cooperatives should be formed to either produce a plan using existing county or municipal staff or private contractors.

A regional plan or program is similarly problematic, but involves a much larger spatial scale (region versus watershed). We believe that the conservation strategy for nonfederal lands proposed in this document will be most effective if it is integrated with Federal aquatic conservation strategies including the Northwest Forest Plan, PACFISH, and INFISH. All of these programs would be enhanced if they were linked with one another and with other Federal, State, and Tribal entities into a comprehensive regional salmonid conservation program.

We believe a salmonid conservation program for nonfederal lands will be most effective if it combines both voluntary and regulatory components. To the degree possible, the Agencies should work closely with landowners to mutually identify issues of concern, identify options or guidelines, and provide individual landowners sufficient information to employ protective actions voluntarily. However, given the current status of many salmonids in the Pacific Northwest and past failure of voluntary programs, a regulatory component will be necessary. The establishment of science-based criteria and best-management practices directed at minimizing ecological impacts are important aspects of such an approach.

11.4.2 Monitoring Conservation Efforts Locally and Regionally

The question of who should monitor salmonid conservation activities involves several issues. Individual conservation plans niust be reviewed for adequacy prior to implementation. Once an HCP has been implemented, it must be monitored to ensure that all required provisions have been followed and that it is having the desired effect on salmonid ecosystems. In addition, the process of developing HCPs must be monitored to assure quality and regional consistency.

Review of individual conservation plans should be conducted by Agency staff, technically trained in the disciplines of geology, hydrology, soil science, aquatic ecology, fisheries ecology, and if appropriate, toxicology and engineering. If the watershed or watersheds affected by the HCP contain only nonfederal lands, we recommend at least bi-agency review. If the affected watershed drains contiguous Federal lands, the appropriate Federal land-management agency should be included in the review process. A goal of these reviews should be the development of consistent plans, at least within ecoregions, and regardless of ownership. For HCPs prepared pursuant to the Endangered Species Act, a formal public comment period is required before approval. We also recommend that the overall conservation program itself undergo periodic peer review, with reviewers representing other agencies, academia, and the private sector.

Implementation monitoring for HCPs should mostly be conducted by Agency staff (or by contractors) because HCPs are a contract with the Federal Government. Ideally, persons conducting the HCP reviews will also perform some of the implementation monitoring, especially site inspections. Where remote sensing is involved, staff should include geographers and landscape ecologists with skills in GIS analysis and interpretation of aerial photographs.

Development of a regional assessment monitoring system for salmonid ecosystems is also clearly an Agency responsibility, although with appropriate coordination it could include other Federal, State, Tribal, and private entities. The same is true for monitoring individual HCPs. All three types of monitoring information should be entered into an Agency computer database to facilitate rapid, quantitative analysis.

11.4.3 Additional Issues in Implementing a Salmon Conservation Strategy

We see a clear need for a cooperative Federal, State, and Tribal effort in developing a computer database (with GIS capabilities) to support salmonid conservation planning. Database managers, computer programmers, and statisticians will be needed to ensure effective and responsive operation. Inforination must be readily available to all interested agencies, landowners, and concerned public.

Several issues relating to equitable treatment of landowners also warrant discussion. In attempting to develop a sound ecosystem approach to conservation

on nonfederal lands, the potential exists that landowners who have been good stewards may be asked to restrict activities in certain areas to protect critical salmonid habitats—habitats that are important precisely because the land was well managed—whereas landowners who have intensively and extensively exploited resources may avoid such restrictions. Similarly, where many landowners are involved in a conservation agreement, the actions of one landowner may adversely affect all landowners within the basin. This is an especially important issue in comparing restrictions applied to forest lands with those for urban, agricultural, and range lands. These issues will be difficult to resolve. Alternative conservation trade-offs, land exchanges, tax breaks, or other incentives may provide means for rewarding good stewardship. Conversely, removal of Federal subsidies or other disincentives may be required *to* penalize poor stewards. Finally, we believe it is important that a regional habitat conservation strategy for salmonids consider other factors directly influencing salmonid populations (hatcheries, salmonid harvest, dam operations), as well as the root causes of environmental deterioration (i.e., population growth, resource consumption).

These recommendations acknowledge that ecosystem management will be accomplished through many individual and independent actions. But they also acknowledge that if ecosystem management and salmon conservation are to succeed, each independent action must be integrated into a comprehensive program with a regional conservation objective. The science underlying landscape management and salmonid conservation constantly progresses: thus, implementing **an** effective strategy requires adapting to new information as it is developed. It is our belief that the planning elements contained in this document provide a foundation from which to build a successful strategy by applying what we already know about ecosystem function as well as by facilitating the collection of information that will allow us to improve planning efforts in the future.

12 Purpose

Much as the loss of Pacific Northwest salmon has come from "multiple compounding human acts of commission and omission," their restoration will come from multiple sources and solutions; single-minded pursuit of one or two strategies will ensure failure.

Ellen W. Chu and James R. Karr
Editors' Note in *Illahee,* 1994

Without question, the complexity of social and ecological issues encompassed by the current salmonid crisis exceeds that for any other resource issue in the history of the Pacific Northwest. The widespread decline of salmonids in the region, as Chu and Karr note, is the result of numerous human activities, including land management (logging, grazing, agriculture, mining, and urbanization), water use (hydroelectric operations, irrigation withdrawals, domestic consumption, dilution of industrial and domestic effluent, and river transportation of commodities), and fishery management (harvest, hatchery supplementation, and introduction of non-native species). The direct and indirect linkages between the health of salmonid populations (and aquatic ecosystems in general) and these many industries and activities have important implications. Restoration of salmonids will affect virtually everyone who resides in the Pacific Northwest through 1) costs of water, food, electricity, and other commodities; 2) the availability of jobs in the fishery, forest, agricultural, and mineral industries; 3) restrictions on use of private lands; and many other avenues. Furthermore, the development of successful restoration strategies will require an unprecedented level of cooperation among managing agencies and between the public and private sectors.

Recent listings of anadromous salmonid stocks in Washington, Oregon, and California under the Endangered Species Act (ESA) have prompted a number of private and other nonfederal landowners to prepare habitat conservation plans (HCPs) pursuant to Section 10 of ESA. ESA allows for incidental take of threatened or endangered species (see Section 9.3 in Part I for a definition of "take") or modification of their habitats provided that a habitat conservation plan is developed by the applicant and subsequently approved by the Secretaries of Commerce (anadromous species) or Interior (resident species). In addition, a number of watershed councils have formed in the Northwest for the purpose of developing conservation strategies for salmonids on nonfederal lands or lands of mixed Federal, State, Tribal, and private ownership. The National Marine Fisheries Service, Fish and Wildlife Service, and U. S. Environmental Protection Agency (the Agencies) seek to develop a coordinated program for evaluating habitat conservation plans, prelisting agreements, and other conservation efforts on nonfederal lands to assure compliance with **ESA,** the Clean Water Act, and other relevant legislation.

In the remainder of Part II, we develop an ecosystem-oriented approach to the planning and monitoring of salmonid habitat conservation efforts on nonfederal lands in the Pacific Northwest. We focus on the effects of land- and water-use practices on salmonids and their habitats and on how these impacts can be minimized through improved planning and management. Although habitat degradation is clearly a major cause of salmonid declines across much of the Pacific Northwest, many salmonid populations will likely continue to decline regardless of how well the landscape is managed unless steps are taken to reduce other human impacts (e.g., overharvest, hatcheries). Thus, the recommendations contained herein should be considered only part of a larger, comprehensive salmonid restoration strategy.

We intend for this document to provide 1) a conceptual framework from which the Agencies can organize a regional conservation strategy, 2) practical information for nonfederal entities to assist them in preparing HCPs and other salmonid conservation plans, 3) guidelines for monitoring HCPs and other conservation efforts, and 4) criteria by which the Agencies can evaluate habitat conservation activities. These four elements are presented together because it is essential that landowners or watershed councils preparing HCPs or other conservation plans have a thorough understanding of the Agencies' management

goals (and the reasoning behind those goals) if watershed- and site-level conservation efforts are to succeed. In addition, information in this document may assist county and local governments in developing zoning regulations, land-use ordinances, development standards, and other regulations or guidelines that are compatible with salmonid conservation objectives.

Chapter 13 presents several broad ecological goals that should guide regional salmonid conservation efforts, emphasizing the role of nonfederal lands in achieving these regional goals. Chapter 14 outlines specific planning elements that should be incorporated into habitat conservation plans for nonfederal lands. "Planning elements," as used in this document, comprise three parts: 1) identification of issues and concerns at site, watershed, basin (or provincial), and regional levels; 2) specific evaluations needed to determine if proposed activities are likely to disrupt watershed processes, aquatic ecosystems, salmonid species, or other biota; and 3) data or information needed to perform these evaluations. Incorporated into this section is an evaluation of the effectiveness of current Federal and State forest practice rules for Washington, Oregon, California, and Idaho in protecting riparian functions. Chapter 15 proposes a monitoring strategy to ensure that habitat conservation plans are both implemented and produce the desired outcome. In Chapter 16, we suggest a framework for implementing this conservation strategy. The volume concludes with an Appendix listing sources of data and information that landowners and agencies may find useful in developing and evaluating HCPs and other conservation efforts.

The recommendations made in this document are intended as guidelines for conservation planning, not formal requirements. Each planning situation is likely to be unique, and not all planning elements may be warranted in each case. Nevertheless, a conservation strategy for nonfederal lands will be most successful if it fosters consistency among conservation planning efforts, builds on or complements existing programs that promote ecosystem management, and integrates into a broader regional recovery program for both Federal and nonfederal lands. Succeeding in this effort will require close coordination and cooperation among Federal, State, Tribal, and local governments, and the private landowners or watershed councils who engage in salmonid conservation efforts.

13 Goals of Salmonid *Consewation*

An effective restoration strategy to ensure the long-term persistence of salmonids must be grounded in principles of watershed dynamics, ecosystem function, and conservation biology (Frissell 1993). Part I of this document was intended to provide the technical foundation from which such a strategy could be developed. We presented a detailed discussion of physical, chemical, and biological processes operating upon the landscape, within riparian areas, and within aquatic ecosystems that influence the ability of these ecosystems to support salmonids. We also discussed how land-use activities alter salmonid habitats by disrupting these natural processes, particularly the rate of delivery of water, sediment, fine and coarse organic debris, and dissolved substances to streams, rivers, lakes, and estuaries.

From this review of the literature, we have identified five ecological and biological goals as central to salmonid conservation:

- Maintain and restore natural watershed processes that create habitat characteristics favorable to salmonids. It is essential that whole, contiguous landscapes be managed to protect natural *processes* (i.e., the natural rates of delivery of water, sediment, heat, organic materials, nutrients, and other dissolved materials), rather than specific *states* (Reeves et al. 1995). Ecosystems are dynamic, evolving entities that must be managed to retain their capacity to recover from natural disturbances (e.g., climate change, fire, disease, floods). Active, in-channel habitat restoration is recommended only for severely degraded systems where failure to act may cause irreparable harm to the aquatic system; such restoration should be an interim measure, not a measure to mitigate damage to streams and riparian areas or to exempt them from protection (FEMAT 1993).

  In stating that an important goal of salmonid conservation is to maintain and restore natural processes, we recognize that an expectation of returning ecosystems to pristine conditions is unrealistic, particularly on private lands, given the current degree of human disturbance to the landscape and the continued demand for other natural resources. Nevertheless, substantial progress toward the goal of naturally functioning aquatic ecosystems and salmonid habitats can be made by identifying portions of the landscape where the signatures of key processes are strongest and employing management practices that are appropriate for the level of sensitivity. Important in this approach is considering how multiple activities, in space and time, interact to influence salmonid habitats.

- Maintain habitats required by salmonids during all life stages—from embryos and alevins through adults—and functional corridors linking these habitats. The complex life histories of salmonids frequently demand a wide array of habitat types. Different portions of a watershed may accommodate spawning and rearing, and these habitats vary with species. Large lowland rivers are rearing habitats for some species and serve as important migration corridors through which anadromous fish pass on their way to and from the sea. These migration routes must be ecologically healthy with high water quality, the physical attributes required for holding, feeding, or hiding, as well as the biological elements favorable to salmonids during these physiologically demanding transition periods.

- Maintain a well-dispersed network of high-quality refugia to serve as centers of population expansion. Conservation biologists suggest that the most fundamental goal of species (and ecosystem) protection is to preserve those habitats that retain a high degree of ecological integrity. Populations within these "healthy" habitats have the greatest probability of surviving natural disturbance events or long-term shifts in environmental conditions.

- Maintain connectivity between high-quality habitats to allow for reinvasion and population expansion. The high degree of landscape fragmentation that has resulted from human activities has left many salmonid populations in relative isolation. Long-term persistence of salmonid metapopulations depends on developing connectivity between subpopulations through restoration and maintenance of corridors, so that these populations can interact in a natural fashion.

- Maintain genetic diversity and integrity within and among salmonid stocks and species. Preserving natural genetic diversity at the level of

individuals, stocks, and species enhances the ability of salmonids to respond to and survive natural environmental change, as well as human-caused perturbations. The loss of life-history types or stocks diminishes the ability of salmonids to persist over the long term. Wild salmonid stocks are subtly adapted to local environmental conditions; alteration of the genetic integrity of these stocks through planting of hatchery fish, exploitation, construction of barriers, or other means renders them less adapted to their environments,

We believe that these ecological goals for attaining regional recovery of salnionids should underlie conservation efforts at all levels, from site-specific prescriptions to watershed, basin, and regional plans. Activities that maintain natural watershed and ecological processes, facilitate the expansion of refugia, enhance connectivity between refugia or from headwaters to the ocean, and allow full expression of the genetic potential of the species should be encouraged; those activities that do otherwise should be discouraged.

The National Marine Fisheries Service has indicated that Federal lands and Federal activities shall bear as much of the burden as possible for conserving listed salmonid populations and the ecosystems upon which they depend (NMFS 1995a, b). Yet nonfederal lands account for more than 50% of the total land area in the Pacific Northwest and they include some of the most productive waters; consequently, conservation on nonfederal lands must be an integral part of a regional salmon recovery program. The goals listed above cannot be met entirely on Federal lands for a number of reasons. First, the wide range of habitats demanded by the complex life histories of anadromous salnionids cannot be provided on Federal lands alone. Second, persistence of salmonids requires preservation of genetic and life-history diversity of salnionid stocks across the landscape; loss of salmonid stocks on private lands diminishes the overall capacity of the species to persist in the face of natural environmental change (e.g., climatic shifts). Third, connectivity between relatively intact refugia on Federal lands can be maintained or restored only by conserving ecologically healthy corridors on nonfederal lands. And fourth, many of the most productive salmonid habitats once occurred in low-gradient river reaches and estuaries, areas that are largely in private or municipal ownership; consequently, recovery of salmonids to healthy or fishable levels will require restoration of these biologically important waters.

In addition to the ecological goals discussed above, habitat conservation on private lands should consider important societal goals as well. The harvest of salnionids is an integral part of many cultures in the Pacific Northwest. Salmonids have significant ceremonial and economic importance to Native American cultures of the region. Furthermore, anadromous salmonids have until recently supported tens of thousands of commercial fishers along the coasts of California, Oregon, and Washington. Sport fishing provides an important source of recreation and food as well as diversifying local economies in the region. The loss of salmonid stocks in Pacific Northwest rivers diminishes the rich cultural heritage unique to this region. In addition, the local economies of many small communities in the Northwest are based on the use or extraction of natural resources. Conservation activities may affect the ability of private landowners to continue to extract commodities to sustain their livelihoods. Support for conservation programs by private landowners is essential for attaining the ecological goals outlined above.

Because many private landowners and niunicipalities have been or are likely to be affected by ESA listings, this document focuses on HCPs; however, the recommendations are equally appropriate for other habitat conservation activities intended to allow owners of nonfederal lands to proceed with land-use or water-use activities while satisfying endangered species, clean water, or other legal mandates. As noted above, many ecological and social issues related to salmonid conservation involve region- or basin-level considerations. Private landowners, municipalities, States, or other nonfederal landowners should be made aware of these considerations, but cannot reasonably be expected to technically address all of these concerns within a conservation plan. Thus, the recommendations provided in this document are intended to aid both owners of nonfederal lands engaged in conservation planning and the Federal agencies responsible for administering HCPs and broader conservation programs.

Finally, we emphasize that the process of developing and approving habitat conservation plans should be an evolving one. Limits to scientific knowledge have precluded us from making specific recommendations on many aspects of conservation planning. Furthermore, specific criteria proposed in this document may not be appropriate in all circumstances, owing to the inherent variation in aquatic ecosystems across the landscape and at any one location through time. Consequently, these criteria should be viewed as indicators of ecosystem or habitat condition, not rigid standards. New information, some of which may be gained through the extensive monitoring strategy suggested herein, should be incorporated into the process as these data become available. The specific planning and monitoring elements proposed in this document represent our best professional judgement. Review and revision of this document among the Agencies, the scientific community, and the public is essential to further develop credible restoration strategies.

14 Planning Elements

Ecosystem-oriented approaches to land and resource management are being recommended by scientists and the management agencies that oversee activities on public and private lands (FEMAT 1993; FS and BLM 1994a, 1994c). Although the term "ecosystem management" has been defined a number of different ways in the literature, the goal of preserving ecosystem integrity while deriving sustained benefits for human populations is common to most definitions (Montgomery et al. 1995). Ecosystem management represents a substantial departure from historical management approaches that 1) attempted to maximize the efficiency with which a limited number of commodity values were extracted or developed, 2) focused on single species rather than on biological communities or assemblages, and 3) were based on administrative units or areas of single ownership rather than on more ecologically meaningful units, such as watersheds, basins, and ecoregions.

As a society, our thinking about applied ecosystem management is in the early stages. Not only is our scientific understanding of ecosystem processes incomplete, but our current institutional structure—with responsibilities for resource protection and production fragmented among various Federal, State, and local agencies—can make regional ecosystem planning difficult. Although there are encouraging signs of greater interdisciplinary research and interagency cooperation, the development of regional strategies for salmonid conservation will be an ongoing and evolving activity for decades. Nevertheless, society can begin working immediately toward larger ecosystem goals by implementing sound management practices at the scales of watersheds and local sites. Habitat conservation ecosystem-oriented approaches to land management plans (HCPs) offer an opportunity to begin to integrate habitat conservation efforts on nonfederal lands with similar efforts on Federal lands.

Section 10 of the Endangered Species Act (ESA), which specifies conditions permitting the incidental take of species, contains several key provisions that are designed to ensure that the intent of the Act is realized and that reasonable alternatives to the proposed activities are considered. Specifically, ESA requires that HCPs address

1) the impact that will likely result from the taking (of a species or its habitat),
2) the steps that the applicant will take to minimize and mitigate such impacts and the funding that will be available to implement such steps,
3) the alternative actions to such taking that the applicant considered and the reasons why such alternatives are not being utilized,
4) such other measures that the Secretary may require as being necessary or appropriate for purposes of the plan.

To satisfy the intent of ESA, it is important that HCPs be developed and evaluated within the context of larger ecosystem restoration strategies. In this regard, a broad spectrum of issues should be addressed in HCPs: site-specific impacts; cumulative effects of multiple activities (in space and time) throughout a watershed; the distribution and status of salmonid species or population segments at region, basin, watershed, and site levels; and the status of other biota and resource values. Montgomery et al. (1995) and the *Federal Ecosystem Analysis Guide* (REO 1995)[1] suggest that implementation requires a hierarchy of planning scales, including regions,

[1] The Federal watershed analysis protocols were first published by the Regional Ecosystem Office (REO) under the title *Federal Agency Guide for Pilot Watershed Analysis* (REO 1994), which we hereafter refer to as the *Pilot Watershed Analysis Guide.* A revised version, *Ecosystem Analysis at the Watershed Scale: The Revised Federal Guide for Watershed Analysis, version 2.1* (REO 1995), contains additional analytical modules as well as revised protocols for existing modules. In this document, we cite the revised guide under the shortened name *Federal Ecosystem Analysis Guide;* however, because the revised version is supplemental to the original, the reader should obtain both of these documents.

basins or provinces (i.e., groups of smaller basins with similar characteristics, such as small coastal streams in Oregon, Washington, or California), watersheds, and individual sites (Figure 14-1). Watershed analyses and site prescriptions that are the most likely components of conservation plans should be imbedded within analyses at larger spatial scales. As suggested earlier, most planning activities at the regional and provincial levels are beyond what can reasonably be expected of private landowners developing HCPs; thus responsibility will fall primarily on State and Federal management agencies for ensuring that HCPs for sites or watersheds satisfy larger ecosystem restoration goals.

The hierarchical approach for conservation planning on public lands suggested by Montgomery et al. (1995), FEMAT (1993), and the *Federal Ecosystem Analysis Guide* (REO 1995) is both necessary and appropriate for protection of salmonids on private lands as well.

> *Each of these scales of analysis and planning are necessary for implementing ecosystem management because: (1) the distribution and environmental requirements of a number of species are not organized on a watershed basis, and thus need to be considered across levels of the analysis and planning hierarchy; and (2) the spatial context within which the watershed lies is an important factor in evaluating the ecological significance of land management alternatives.*
>
> (Montgomery et al. 1995)

In the remainder of this chapter, we identify what we believe to be key planning elements at various levels in the planning hierarchy that should be involved in the preparation and evaluation of HCPs or other conservation efforts. Again we note that this list of planning elements is purposely broad to cover a wide range of activities and conservation issues; the specific elements to be considered in an HCP or other conservation plan will vary depending on the specific activity proposed and relevant ecological issues.

14.1 Region and Basin Levels

14.1.1 Key Issues

This chapter identifies issues and analyses to be conducted at the scale of regions and basins (or provinces) to determine whether watershed- and site-level conservation efforts will facilitate attainment of regional conservation goals outlined in Chapter 13 of this document. As stated earlier, analyses at these scales will be conducted by Federal, State, and Tribal agencies. Typically, basins and provinces encompass areas of thousands to tens of thousands of square miles, e.g., the Willamette, Deschutes, Yakima, Clearwater, and other major sub-basins of the Columbia River system, and the Smith and Eel Rivers of northern California. Several biological and ecological issues are relevant at these large spatial scales. Biodiversity, species or stocks at risk, cumulative effects, habitat fragmentation and connectivity, metapopulation dynamics, and total salmonid production are all issues that transcend watershed boundaries and thus cannot be satisfactorily addressed without basin-level and regional assessments. Similarly, issues related to the estuarine and marine environments into which anadromous salmonids enter also need to be addressed at this level (e.g., pollutants, sediment loading, fish harvest management) because the perpetuation of populations within a watershed or basin depends on maintaining these habitats and limiting mortality from fishing.

We believe three initial steps by the managing agencies are essential to address these issues: 1) establish a network of key watersheds on private lands that complements Federal key watersheds designated in FEMAT (1993) for westside ecosystems and those currently being developed for eastside systems (FS and BLM 1994b); 2) adopt riparian protection standards for all riparian areas across the four-State area (we do not imply that uniform standards would be appropriate); and 3) delineate evolutionarily significant units (ESUs) for all species of anadromous and resident salmonids. FEMAT (1993) recommends that key watersheds should include watersheds that currently contain habitats or water of high quality, that in the future could provide high-quality habitats, or that are currently habitats for at-risk stocks. We suggest additional criteria: include watersheds with high biodiversity (fish and nonfish species), watersheds that have unique attributes that favor salmonids (e.g., biological "hot spots"), and watersheds or corridors that are important for linking existing refugia, as proposed by the Oregon Chapter of the American Fisheries Society (Henjum et al. 1994). Minimum riparian-protection standards are desired because human activities within the riparian zone have the most direct and damaging effects on salmonids. Protection of the riparian zone is essential for protecting many stream processes, moderating the influence of upland management on aquatic systems, re-establishing connectivity between habitat fragments and biotic refugia, and maintaining ecologically functional migration corridors from headwater streams to the ocean. The delineation of ESUs is needed to clarify biodiversity objectives that in turn should be considered in the establishment of key watersheds. NMFS is in the process of defining

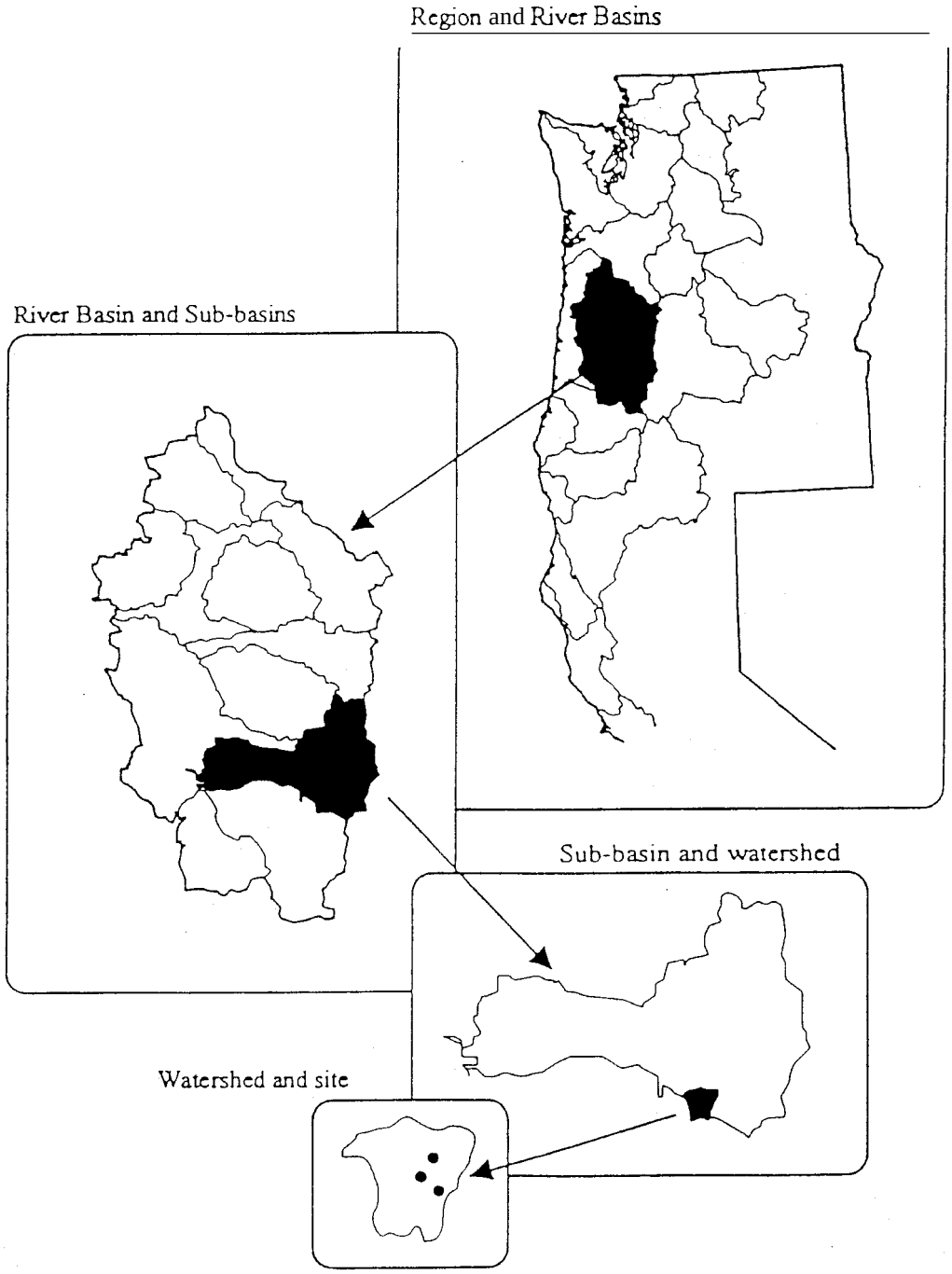


Figure 14-1. A spatial hierarchy for salmonid conservation planning. From REO (1994).

ESUs for all of the anadromous salmon and for steelhead and sea-run cutthroat trout, as part of region-wide status reviews (NMFS 1994). Similar analyses are needed for resident salmonid species, particularly bull trout.

14.1.2 Evaluations

In evaluating habitat conservation plans, the Agencies need to address a series of questions at the region and basin scales that relate to the conservation goals suggested earlier. These are listed by issue below.

Biodiversity

- Is the basin or province an area of high diversity for fish species or stocks, other aquatic species, or terrestrial biota?
- Does the basin possess unique physical attributes that would suggest corresponding unique biological attributes that may not have been identified?
- Does the basin contain narrow endemic populations, or populations with unique genetic or life history attributes?
- Do hatchery populations of salmonids threaten the integrity of wild stocks?

Stocks or Species At Risk

- Are there threatened, endangered, or other at-risk species or stocks in the basin that would be affected by the proposed activities?
- If at-risk stocks are not present in the watershed but present in the basin, could the proposed activity limit the expansion or recovery of at-risk stocks?

Connectivity and Metapopulations

- What is the watershed's location relative to key watersheds on Federal lands? Is it immediately adjacent to or linked to key watersheds on Federal lands?
- Is the area or part of the area covered under the conservation plan used as a migration corridor by anadromous stocks?
- Is the area a potential dispersion corridor for resident stocks?
- Are existing salmonid populations likely "seed sources" for recolonization of degraded habitats?
- Are there physical, chemical, or biological barriers that prevent or inhibit movement of fish to and from the basin?

Salmonid Production

- What is the current importance of the stream or watershed in the overall production of wild salmonids in the basin and region?
- What was the historical importance of the stream in the overall production of salmonids?
- Does the basin have high or low restoration potential?
- Does the area affected by the proposed activity contain any biological "hot spots" (i.e., reaches that support a disproportionate number of fish relative to surrounding reaches)?

Cumulative Effects and Fragmentation

- What are the primary stressors in the watershed? Are these stressors of natural origin or a consequence of human activities?
- Are the proposed activities addressed by the conservation plan likely to exacerbate or mitigate these stressors?
- Would the proposed activities result in further fragmentation of aquatic habitats, thereby diminishing prospects for recovery?
- What are the anticipated future developments (e.g., urbanization and water development) in the basin and region?

Estuarine and Marine Environments

- What are the primary stressors affecting salmonids in the estuarine and marine environments?
- Would the activities proposed in the conservation plan exacerbate or alleviate those stressors?
- What role is fish harvest playing in the health of the affected populations?

We fully recognize that information constraints may prevent many of the above questions from being answered satisfactorily. Frequently, the data for performing these analyses may not exist. (One objective of the regional monitoring program proposed in Chapter 15 of this document is to address these gaps in our information base.) For other issues, data may exist but are not readily available to the managing agencies (see Appendix **A).** The time and cost of acquiring and interpreting these data for analysis of individual conservation plans is almost certain to be prohibitive. Therefore, it is essential that a centralized database be developed containing information relevant to salmonid conservation at the basin or regional level in useful forms, such as maps of species distributions, land-use patterns, water withdrawals, and barriers to migration (physical, chemical, and biological). Development of a regional database management system in support of salmonid conservation planning must be a cooperative effort with other Federal, State, and Tribal agencies. These issues are discussed at length in Chapters 15 and 16.

14.2 Watershed Level

Watersheds with areas of approximately 20–200 square miles are generally the most practical for planning and analysis (FEMAT 1993; Montgomery et al. 1995). It is at this level that linkages between physical and biological processes can be addressed most effectively. In Part I of this document, we identified numerous physical arid chemical processes that affect salmonids and their habitats, as well as biological processes that may be altered by changes in physical-chemical habitat characteristics. Important physical-chemical processes include morphological development of stream channels, sediment transport, hydrology, heat transfer in streanis, nutrient cycling, and various functions provided by standing or downed riparian vegetation (e.g., bank stabilization, sediment control, shading, coarse and fine organic inputs, microclimate, physical structure, etc.). Important biological considerations include the physiological and biological requirements of individual fish (e.g., food, space, migration routes), population-level processes (e.g., local adaptation, life-history patterns and diversity), and community-level interactions (e.g., predator-prey, competitor, and disease-host relationships). As reflected in the ecological goals outlined in Chapter 13, maintaining these processes within the natural range of variability should be a primary goal of watershed-level planning.

The concept of "watershed analysis" evolved out of concern that site-by-site planning of land-use activities has generally failed to adequately address tlie cumulative effects on these complex processes of multiple human activities occurring throughout a watershed. Thus, an important goal of watershed analysis is to assess the potential effects of site-level activities, given tlie historical and projected future patterns of land use, development, and ecological function. In addition to addressing cumulative effects, watershed analysis serves other important functions. It provides an assessment of current conditions within the watershed, which allows existing resource problems to be identified and future activities to be planned in a more ecologically sound manner. Watershed analysis also helps identify specific portions of the watershed that are highly sensitive to human disturbances, such as areas prone to mass-wasting or surface erosion. Climate, soils, geology, topography, vegetation, and many other factors influence how materials and energy are delivered from tlie hillslope to tlie stream channel. Each watershed is unique and will respond differently to land-use practices; thus, no simple prescriptions can be applied uniformly across the landscape to ensure salmonids and their habitats are protected. Watershed analysis allows prescriptions to be developed that account for this inherent variability. Finally, watershed analysis can provide information that helps to refine our understanding of physical and biological processes and how these vary across the landscape. This information can then be used to develop ecoregion- or basin-level standards that more accurately reflect the spatial and temporal variability in ecological processes.

We recommend that watershed analysis be a key component of conservation planning on nonfederal lands. Specifically, we suggest that HCPs and other conservation efforts incorporate evaluations of how proposed activities will potentially affect hydrology (total water yield, peak flow, base flow, and seasonal timing), sediment transport (mass wasting and surface erosion), riparian functions (LWD recruitment, sniall organic litter inputs, stream shading, bank stabilization, and nutrient cycling), channel condition (bed morphology, substrate type, and physical structure), and water quality (stream temperatures and pollutants). Watershed-level analyses should also be conducted to assess biological conditions in the watershed, including fish distributions, habitat condition, and population viability. For HCPs covering other aquatic and terrestrial biota, additional analyses beyond those recommended here would be warranted. The evolving watershed analysis protocols outlined in the *Federal Ecosystem Analysis Guide* (REO 1995) and tlie State of Washington (WFPB 1994) address many critical watershed, riparian, and aquatic processes. The reader is referred to these guides for specific protocols and resulting products.

In each of the sections that follows, we discuss a specific process that may be disrupted by human activities. Each discussion begins with a summary of key issues that we believe should be addressed in HCPs based on our review of the literature (Part I of this document). We then provide recommendations regarding the elements that should be included in the HCP. The intent is to offer general guidelines, not the specific protocols for performing those analyses. The Agencies, and other State, Tribal, and private interests will ultimately need to develop standardized protocols for field sampling and data analyses to ensure consistency among conservation plans to the extent possible. We then present recommendations regarding numeric or narrative criteria that may be used *to* evaluate whether HCPs or other conservation activities are likely to adequately address resource concerns. These recommendations are made only where they can be supported with existing scientific information, Where such data are deficient, we discuss factors that are likely to influence the responses to land management activities so that management agencies have some technical basis for evaluating the adequacy of HCP provisions. We also note that, because of inherent variability in ecological

conditions across tlie landscape and at any one location through time, establishing fixed numeric standards for habitat parameters (e.g., temperature, pool frequency, large woody debris, etc.) may fail to accomniodate this variability and lead to inadequate protections or unwarranted constraints on management activities. Nevertheless, without quantitative ambient criteria, conservation objectives will prove difficult to achieve. Thus we emphasize that where numeric criteria are presented in this report, they are intended to serve as benchmarks or targets. If analysis demonstrates such criteria are inappropriate for the particular region or situation, then these standards should be modified.

An assumption underlying recommendations made in the following sections is that the aquatic habitats affected by the proposed activities support salmonid populations (or influence downstream areas that support salmonids) that are listed, or likely to become listed if not protected, as threatened or endangered under ESA. Consequently, the recommendations are generally conservative in nature and each recommendation may not be appropriate in all circumstances. We also note that because the focus is on salmonids, the recommendations contained herein do not ensure protection of other resource values and in fact may contribute to their degradation. For example, hydrologic effects of timber harvest may be minimized by dispersing numerous small clearcuts over a wide area; however, this may result in a highly fragmented landscape, to the detriment of various wildlife species that would be better protected by employing a few large clearcuts.

14.2.1 Hydrology

Key issues

Land- and water-uses can substantially influence the amount and timing of water delivered to the stream channel. Our review of the literature identified four principal ways in which human activities may influence stream discharge patterns: 1) changes in total water yield; ***2)*** increases in peak flows (particularly during rain-on-snow events); 3) increases or decreases in summer base flows; and 4) altered seasonal timing of flows. In most instances, land-use activities result in an increase in total water yield due to decreases in evapotranspiration demand following the removal of vegetation (Bosch and Hewlett 1982; Satterlund and Adams 1992). However, in one study in the Cascade Range of Oregon, total water yield decreased slightly after vegetation removal, apparently through loss of fog drip (Harr 1982). Increases in peak flows can be caused by tlie reduced evapotranspiration demands (primarily in the fall), changes in the distribution and melting rate of snow, increased efficiency with which water is routed to the stream channel, or any combination of these mechanisms. Summer low flows may increase in response to reduced evapotranspiration demands, but may also decrease in areas where 1) soil compaction reduces infiltration and, hence, subsurface storage, 2) channel incision causes a lowering of the water table (Rhodes et al. 1994), **3)** natural vegetation is replaced with species having greater evapotranspiration demand (Hicks et al. 1991b), and 4) sediment accuniulations in the channel force the stream to flow subsurface. Seasonal timing of flows is affected by many of the above mechanisms, as well as through storage and withdrawal of water for irrigation and hydropower generation.

Stream discharge strongly influences the amount of habitat available to salmonids and the physical characteristics of those habitats; thus hydrologic changes influence salmonids in a variety of ways. Increases in peak flows can scour spawning gravels, change substrate size, redistribute large woody debris within the channel, facilitate channel incision or widening, and accelerate bank erosion. Reduced summer low flows can dewater stream reaches, prevent or inhibit fish migration, and produce higher summer temperatures. Changes in the seasonal timing of flows may disrupt the migration of salmonid juveniles and adults, and may increase the frequency with which disturbances occur during specific life stages (e.g., the incidence of spawning gravel scouring during early fall). In addition, natural flood and drought cycles are important for normal establishment of riparian vegetation. Hydrologic changes in watersheds may indirectly affect salmonid habitats by altering soil moisture content and stability, which affect the rate of sediment delivery to streams via mass failures and surface erosion.

Recommendations

We recommend that HCPs and other conservation plans contain a strategy for preventing cumulative hydrologic effects within the watershed or watersheds. Ideally, the conservation plan should specifically address each of four 'hydrologic issues identified above: total water yield, peak flows, summer low flows, and seasonal timing. For forest, agricultural, and range lands, the following provisions may be appropriate in an HCP: minimizing the areal extent of vegetation disturbance; minimizing the area in hydrologically "immature" condition and deferring further activities until hydrologic recovery has occurred (particularly in areas prone to rain-on-snow events); and minimizing tlie areal extent of roads and skid trails. For urban areas, provisions for minimizing impervious surfaces would be desirable. Where water storage or withdrawals for irrigation or mining occur,

provisions should be included for protecting summer low flows, flows needed for migration, and seasonal flushing flows (flows resembling natural peak flows for scouring substrates), as well as for reducing irrigation withdrawals where inadequate flows are of concern. In most instances, this will involve reducing summer usage and winter storage.

Developing a thorough and defensible management strategy for minimizing hydrologic effects will require a thorough assessment of current watershed conditions. For all land uses, basic information on climate, soils, geology, topography and vegetation will be needed. For forested lands, the analysis should include mapping and assessment of current hydrologic maturity of stands within the watershed; mapping of existing roads, skid trails, landings, and other areas where ground disturbance has occurred; and identifying hydrologically sensitive zones, including areas where rain-on-snow events are likely to occur. Protocols for evaluating potential changes in peak flow niay be found in the Federal *Pilot Watershed Analysis Guide* (REO 1994) and in the Washington watershed analysis guide (WFPB 1994) though modification of the WFPB protocol is under consideration. In California, Sustained Yield Plans may also provide information on potential hydrologic effects in forested systems. For agricultural lands, rangelands, and urban areas, assessment of the current areal extent of disturbance (vegetation, roads, other inipervious surfaces) is also important. The Natural Resource Conservation Service (formerly Soil Conservation Service) has developed hydrologic models that can be used to estimate effects of changes in land use (agriculture, range, and urban) on peak flows in streams (SCS 1982, 1986).

Regardless of land-use type, HCPs should also identify areas where evidence of human-caused hydrologic disturbance exists (e.g., channel incision or widening, dewatering of stream reaches, gullying of incoming drainage channels) and include provisions for mitigating those impacts and reversing the processes that create them to the maximum extent possible. For irrigated agricultural lands, information regarding total withdrawals for the watershed should be presented.

Evaluation Criteria

Our review of the literature found no widely accepted method for determining thresholds for minimizing cumulative hydrologic effects. The probability of significant hydrologic changes resulting from land-use activities generally increases with the percentage of the watershed that has been disturbed (Bosch and Hewlett 1982); however, numerous factors, including climate, vegetation type, soils, geology, land surface form, elevation, and type of disturbance all influence the hydrologic response and confound the ability to predict change. The most frequently used method for assessing cumulative hydrologic impacts is the equivalent clearcut area (ECA) method, the application of which is limited to forested ecosystems. The ECA method involves developing coefficients that express the effects of various forest practices in terms of the equivalent clearcut area that would yield a comparable hydrologic response. The model accounts for site characteristics such as vegetation type, elevation, type of disturbance, and time elapsed since the management activity occurred.

Several recent reviews have found the ECA method to be deficient in many respects (Beschta et al. 1995; Rhodes et al. 1994; Murphy 1995). Rhodes et al. (1994) recommend against using the ECA method because it fails to account for many factors that influence the amount of degradation caused by the disturbance, including proximity to the stream or riparian zone, geomorphic sensitivity, and cumulative affects of other activities, such as grazing and mining. Their objections pertain primarily to using the ECA method alone to determine all cumulative effects (e.g., sedimentation, shade, LWD recruitment), not just hydrologic effects. Beschta et al. (1995) address hydrologic aspects of the ECA approach more directly. They note that although increases in water yield in response to logging are well documented, the assumed correlation between increases in water yield and increases in channel-modifying peak flows has not been firmly established. They also suggest that simple coefficients are inadequate to represent the different mechanisms by which peak flows may be generated (e.g., rain, rain-on-snow, and snowmelt systems), though they acknowledge that because coefficients vary with elevation, these effects may be incorporated into the procedure. In addition to these problems, the hydrologic response to clearcutting depends on the size and distribution of the harvest areas; a few large patch cuts are likely to produce greater increases in yield than many small cuts of equivalent total area. Finally, clearcuts of similar size may exhibit different hydrologic responses depending on the specific harvest and yarding practices and the resulting degree of soil compaction.

Despite these limitations, the ECA method may be useful as a coarse-level indicator of potential hydrologic problems within forested watersheds. Bosch and Hewlett (1982) and Satterlund and Adams (1992) both conducted extensive literature reviews regarding changes in water yield associated with logging and other forest treatments in coniferous forests and found that in most instances, water yield increased if 15%–30% or more of the watershed was disturbed. It should be noted that these reviews

uncovered few studies in which less than 20% of the watershed was disturbed, so changes in yield may occur with less extensive disturbance as well. McCammon (1993 in Murphy 1995) concluded that the level of risk was low in coniferous stands where ECA levels were less than 15% of forest stands less than 30 years in age, but increased at higher ECA levels. His assessment considered several processes in addition to hydrology, and assumed that hydrologic recovery occurs in 15–30 years, Together these observations suggest that no more than 15%–20%of the watershed should be in a hydrologically immature state at any given time. Given the uncertainties associated with the ECA approach, this threshold value should be used only as a general guidepost, not as an absolute measure of cumulative effect. For example, if significant portions of the watershed lie in the transient snow zone, or if past harvest has occurred in hydrologic source areas, more conservative ECA threshold values may be appropriate. Similarly, more conservative measures would be appropriate where channel condition has already been degraded by hydrologic changes, or in watersheds where lack of large wood increases the potential for damage during high flows. In two recent evaluations, the ECA model underestimated changes in total water yield observed in the field (King 1989; Belt 1980 in Reid 1993), underscoring the need to exercise caution in using this method as anything other than a general indicator.

Little information exists regarding possible thresholds of liydrologic disturbance on range and agricultural lands. Although the potential for increased water yields is generally less where precipitation is lower, the greater likelihood for overland flow and rapid routing of water to the stream channel suggest these landscapes may be just as likely to produce channel-modifying peak flows in response to human disturbaiices. Methods analogous to tlie ECA method are difficult to apply for range and agricultural lands because the nature of the land disturbance. On forested landscapes, particularly where clearcutting is the primary harvest method, it is relatively easy to define discrete areas of disturbance. On rangelands, the intensity of disturbance is generally lower than for logging, except perhaps in the riparian zone; however, the areal extent of disturbance is usually high and the alteration persists as long as grazing continues. In agricultural areas, both the intensity and areal extent of disturbance are high and the hydrologic response is confounded by the effects of irrigation withdrawals and storage, as well as differences in evapotranspiration demand of crops compared with natural vegetation. Because of the lack of scientific information, we cannot make specific recommendations regarding thresholds of disturbance for these land uses.

In urban areas, tlie magnitude of peak flows and frequency of high flow events generally increases as a function of the percent area with impervious surfaces (e.g., rooftops, roads, sidewalks, parking lots, etc.). Two studies in urban areas of the Pacific Northwest suggest that increased frequency of peak flows resulting in significant changes in stream channel stability can occur when the percent imperviousness exceeds 10% (Booth 1991; Booth and Reinelt 1993). Other studies have shown decreases in macroinvertebrate diversity (Klein 1979; Steedman 1988; Schueler and Galli 1992; Shaver et al. 1995), fish diversity (Scliueler and Galli 1992), and degradation of fish habitat or declines in abundance (Steward 1983; Shaver et al. 1995) when percent imperviousness exceeds 7%–12%. These changes are not entirely due to hydrologic stress, because pollutants and other factors may also contribute to degradation. Nevertheless, these studies suggest that HCPs developed for urban areas should seek to minimize percent impervious area, preferably below the apparent 7%–10% threshold.

We found no established methodologies for addressing cumulative watershed effects on summer low flows or changes in seasonal timing of stream discharge and therefore cannot recommend watershed-level numeric criteria related to these issues. Landowners should strive to minimize changes relative to natural flow regimes in the drainage. In systems where reduction in summer flow and the resultant higher temperatures may adversely reduce salmonid production or prevent existence, construction of dams or increases in water withdrawals should be avoided.

14.2.2 Sediment Transport

Key issues

Land-use activities substantially alter the rate at which sediment is delivered to streams via both mass wasting and surface erosion processes (reviewed in Chapter 6). Acceleration of mass wasting and surface erosion occurs in response to removal of vegetation or groundcover, disturbance to soils, and disruption of hydrologic processes (primarily changes in soil moisture content and water routing). In disturbed forested systems, mass wasting events (e.g., landslides, debris avalanches, earthflows, bank failures) are the most important source of sediment inputs to streams and most often occur in association with roads, because of failure of cut and fill slopes, stream crossings, and culverts (Furniss et al. 1991). Surface erosion is generally less important on forested lands because of tlie high infiltration capacity of forest soils; however, significant surface erosion may occur in certain geologic types and on

road surfaces, skid trails, landings, and burned or scarified areas where soils are exposed or compacted and where a lack of adequate drainage structures results in channelized surface flows. In grazed and row-crop agricultural systems, the degree of soil disturbance and vegetation removal is typically more extensive than occurs during timber harvest; consequently, the potential for surface erosion is generally greater than on forest lands. In these systems, surface erosion is likely to be a more important source of sediment inputs than mass failures, except on steep terrain or along stream banks.

Increases in sediment delivery to streams can influence salmonids and their habitats in numerous ways. Increased inputs of sediments can result in increased fractions of fine sediments in spawning gravels that niay both reduce intragravel flow of oxygen to developing embryos and entomb alevins. Increased fine sediments may also reduce interstitial spaces in cobble that juvenile salmonids use as winter cover. Large amounts of sediment delivered to streams can effectively reduce pool volume, decreasing rearing habitat for juvenile and resting pools for migrating adults. Elevated sediment loads also increase the frequency of channel scour and fill events, increase channel width through aggradation, and decrease stability of large woody debris. Sedimentation of bottom substrates interferes with the production and diversity of macrobenthos by eliminating rearing space and preventing hyporheic movement. Finally, increases in turbidity and suspended sediments can interfere with normal feeding by salmonids and cause gill damage.

Recommendations

We recommend that HCPs and other conservation plans develop a long-term plan for minimizing cumulative sediment delivery to streams. Important provisions of a conservation plan should include minimizing or avoiding land-use activities (logging, yarding, grazing, farming, mining, road construction) in areas susceptible to mass wasting and surface erosion and in riparian zones; minimizing total road density within tlie watershed, including limited entry to roadless areas; developing a road maintenance schedule to prevent, identify, and mitigate sediment impacts; and active restoration of roads and skid trails no longer in use, particularly those in riparian areas.

Plans for minimizing sediment impacts should be based on a thorough assessment of existing erosion and mass wasting problems within the watershed and their association with specific site conditions and land-use activities. Each watershed contains unique vegetative, soil, geologic, and climatic attributes. Consequently, recognizing specific combinations of characteristics that have led to mass failures or surface erosion in the past provides the best means for identifying areas where risk of future erosion is high. Information on past mass wasting events and surface erosion can be obtained from on-the-ground surveys, aerial photographs, and historical reports. Mapping these areas can assist in developing long-term roading and harvest plans. Analytical approaches for assessing potential for landslides, debris torrents, gully erosion, sheet and rill erosion, bank erosion, and for estimating total sediment yield can be found in the *Pilot Watershed Analysis Guide* (REO 1994) and the Washington watershed analysis guide (WFPB 1994). Additional methods for assessing erosion on forested lands can be found in EPA (1980a) and in Knighton and Soloman (1989). Methods also exist for modeling sediment yields from small watersheds in agricultural and urban areas (Wischmeier and Smith 1978). The goals of these analyses are to estimate the spatial extent of these erosional processes within the watershed, to relate their occurrence to land-use type or watershed characteristics, to assess the resulting delivery of sediment to streams, and to identify areas within the watershed that are high risk for specific types of erosion.

Evaluation Criteria

Because complex interactions among many factors determine the rates of surface erosion and mass failure, it is difficult to develop specific guidelines for determining the adequacy of HCPs or other conservation efforts in relation to erosion. We found little information in the literature that would support the development of numeric criteria for the purposes of preventing cumulative sediment impacts at the watershed level. Nevertheless, tlie relative risk of erosion from **an** area may be assessed based on qualitative and quantitative descriptions of climate, geology, soils, topography, and vegetation. In addition, historical information on landslides in unmanaged or old-growth basins niay offer additional insights into appropriate criteria. In tlie paragraphs below, we discuss specific attributes that have been identified in the literature as important in determining mass soil movement and surface erosion risk.

Mass Wasting. The factors most often associated with mass failures are slope steepness exceeding the angle of internal friction, wet soils, geology and soil texture susceptible to failure, and removal of vegetation.

Slope. Slope gradient is generally the most important determinant of mass failure risk, although critical thresholds for slope vary with the type of mass soil movement. For debris avalanches and

flows (rapid-shallow mass soil movements), Swanston et al. (1980) suggest that risk is high on slopes greater than 34°, medium on slopes between 29 and 34°, and low on slopes less than 29°. Satterlund and Adams (1992) suggest that a critical slope threshold for mass failures under a variety of conditions lies around 30". Based on an extensive review of the literature, Sidle et al. (1985) conclude that most slopes greater than 35" are subject to rapid mass soil movements (i.e., debris avalanches, landslides) and many slopes greater than 25° are susceptible to failure, particularly if the soil niantle is poorly bound to the underlying rock.

Slower mass soil movements, including rotational slumps, earthflows, and soil creep, may occur on more gentle terrain. Swanston et al. (1980) conclude that risks of slumps and earthflows are high on slopes > 30°, medium on slopes from 15-30°, and low on slopes < 15". Sidle et al. (1985) suggest that lower limits for initiation of mass failures are 7–18" for rotational slumps, 4–20" for earthflows, and 1.3-25° for soil creep. They also note that extensive mass soil movements have been observed on gradients of 12–25° in northern California and that these slower movement processes niay contribute more sediments to these streams than rapid failures on steeper slopes.

The above reviews suggest that for all types of mass soil movements, the risk of mass soil movements is high on slopes > 30"; we therefore recommend that activities be minimized or avoided on slopes exceeding this gradient except where a slope stability assessment conducted as part of a watershed analysis indicates the risks of mass wasting and delivery of material to stream channels is low. For lesser slopes, risks of mass failure niay also be high and final decisions regarding appropriate land management practices should be based on site-specific analyses of precipitation and hydrologic characteristics, soil type, geology, and other site conditions discussed below. In general, increasingly conservative standards should be adopted with increasing likelihood that sediments generated by mass failures will enter the stream channel.

Soil Moisture. The risk of mass failure typically increases as soil moisture increases. **As** soils become saturated, positive pore water pressure exerts force that can allow shear stress to overcome resistant forces of cohesion, friction, and binding strength of roots. Consequently, the probability of mass failures increases with intensity of precipitation. Satterlund and Adams (1992) suggest that landslide hazard increases substantially when storm precipitation exceeds 12.5 cm (4.9 in), but note that less intense storms can trigger landslides when soils are already wet from previous precipitation events. Swanston (1991) suggests that critical rainfall intensities for debris avalanches lie between 7.6 and 15.2 cm (3–6 in) in a 24-hour period. Other types of mass failures, including slumps and earthflows, depend more on long-term water accumulation (seasonal and annual) than on individual storm events. Swanston et al. (1980) concluded that risks of debris avalanches and debris torrents are high for areas receiving more than 203 cm per year (80 inches per year) total precipitation or 102 cm per year (40 inches per year) distributed over a clearly defined rainy season, moderate for areas receiving between 51 and 102 cm per year (20–40 inches per year), and low for areas with less than 51 cm per year (20 inches per year). Thus, both the potential for high-intensity rainfall events (or rain-on-snow events) and total annual precipitation should be weighed when evaluating mass failure risk.

Landform and subsurface drainage characteristics also influence the relationship between soil moisture and the likelihood of mass failure. Convex slopes tend to disperse water, whereas concave slopes concentrate water into smaller areas, facilitating rapid, localized increases in soil moisture during storms (Sidle et al. 1985). In addition, because water tends to drain both downward and laterally towards the stream channel, soil moisture tends to be highest towards the base of slopes and near the stream channel. Landslide risks are also high where the density of drainage depressions is great. Risk also increases where bedrock or other impervious materials underlie a shallow soil mantle (Swanston et al. 1980), which causes subsurface waters to concentrate. The presence of permeable low-density zones above impervious layers indicates saturated flow parallel to the slope, which confers a higher risk of hillslope failure. Springs on hillslopes are also indicative of near-surface flow. More conservative land management is warranted on slopes exhibiting on or more of these characteristics.

Geology and Soil Type. The geologic factors that tend to predispose hillslopes to various kinds of mass failures are generally well known (Sidle 1985). Shallow, rapid mass soil movements (e.g., debris avalanches and torrents) are typically associated with one or more of the following conditions: shallow soils overlying hard, impermeable surfaces; parallelism between the slope and underlying planar rock structures (bedding planes, fractures, joints, and faults); and unconsolidated or weakly consolidated soils. Earthflows, slumps, and soil creep occur most frequently where soft, clay-rich rocks form a thick, plastic soil mantle (Swanston et al. 1980; Sidle et al 1985; Satterlund and Adams 1992). Like debris avalanches, these slower moving failures are also

more likely where underlying planar structures run parallel to the hillslope.

Specific soil textures that influence susceptibility of hillslopes to debris avalanches and torrents have been summarized by Swanston et al. (1980). They conclude that risk of debris avalanches is high for unconsolidated, noncohesive soils and colluvial debris, including sands and gravels, rock fragments, weathered granites, pumice, and noncompacted glacial tills with low silt content (< 10%) and no clay. They suggest that the risk of failure is intermediate for unconsolidated, noncohesive soils and colluvial debris that have moderate silt content (10%–20%) and low (< 10%) clay content, Fine-grained, cohesive soils with greater than 20% clay or mica content are considered low risk soils for rapid mass failures. Soil texture depends on interactions between parent rock type and climatic conditions. Siltstones, shales, mudstones, pyroclastics (volcanoclastics), and serpentines generally weather rapidly into clays; consequently, soils derived from these materials may be less prone to sliding. In contrast, soils derived from granites and sandstones are typically shallow and cohesionless and, therefore, more susceptible to landslides and debris avalanches (Satterlund and Adams 1992)

Soil texture is also a critical factor in regulating earthflows and slumps, although the characteristics that result in slumps and earthflows differ from those typically associated with rapid mass soil movements. Swanston et al. (1980) summarize the relative risks of slumps and earthflows in relation to soil texture as follows. They suggest that fine-grained, cohesive soils derived from sedimentary rocks, volcanics, aeolian and alluvial silts, and glaciolacustrine silts and clays are prone to slower earth movements. In addition, soils with high clay content (>20%) or with clay minerals that swell upon wetting (e.g., the smectite group) also are at relatively high risk of earthflows and slumps, as are the amorphous clays (Satterlund and Adams 1992); thus, soil types associated with slumping may differ from those that lead to more rapid mass movements. Soils of variable texture with both fine and coarse grained components arranged in layers or lenses, and soils with clay fractions derived from illite and kaolinite groups are at medium risk of slower mass movements. Soils with variable texture and low or widely dispersed clay fractions generally have low risk of failure.

Vegetation Removal. The removal of vegetation influences mass failure processes in two ways. First, the reduction in evapotranspiration demand increases the amount of water within the soil, which may elevate soil water tables (Chamberlain et al. 1991). Second, the root network of vegetation niay help stabilize soils by creating a laterally strong matrix of roots and soil, by anchoring the soil mantle to more stable underlying rock or soil, and by providing local reinforcement in the immediate vicinity of trees (Sidle et al. 1985). As roots decompose following logging, these stabilizing effects diminish. It is unclear which of these mechanisms is most important in stabilizing soils (Sidle et al. 1985) and, consequently, it is difficult to make recommendations regarding management practices related to vegetation removal. Nevertheless, in areas with shallow soils and steep slopes, retention of both large conifers and deciduous understory is advised. Procedures for assessing root strength influence on landslide risk are available (see Sidle et al. 1985).

Swanston et al. (1980) note that the size and location of timber harvest units, as well as subsequent land treatments, can greatly influence the incidence of debris avalanches and torrents, as well as earthflows and slumps. They suggest that large clearcuts that create continuous downslope openings have higher risk of failure than smaller patch cuts (< 20 acres) or partial cuts because of the combined effects of increased soil moisture and, for shallow slides, reduced root strength. They also suggest that failure risk can be reduced by avoiding post-harvest broadcast burning on sites with slopes > 34".

Surface Erosion. The vulnerability of areas to surface erosion depends on several site characteristics including slope, soil type (infiltration rate and degree of compaction), drainage characteristics, and the presence of vegetation or organic litter.

Slope. The erosive force of water increases with the velocity of runoff; consequently, the rate of surface erosion increases with both the gradient and length of the slope. EPA (1980a) reported that soil loss increases approximately as the 1.4 power of percent slope for slopes less than 20% (11°) and as a power of slope length that increases with gradient. On rangelands, Heady and Child (1994) state that a doubling of the slope doubles the erosive power of water and increases the amount of material eroded by a factor of 16. Consequently, incremental increases in slope gradient result in a disproportionately greater risk of surface erosion. We found no published reference to critical slope thresholds for minimizing surface erosion. However, Henjum et al. (1994) concluded that in order to control sediments on eastside forests, no logging should be conducted on slopes with gradients steeper than 17" on pumice soils (highly erosive) and 31° on other soil types. They also recommended that on slopes between 17" and 31°, 40% of the basal area should be retained with half of this area consisting of trees larger than the mean diameter. These recommendations also offer a basis for agricultural, range, and urban lands

to limit erosion, although the greater frequency and areal extent of disturbance on these lands may call for more protective measures.

Soils. Soil structure and composition are also important factors in erosion. Although there are a large number of soils within the range of Pacific salmon, typical soil types offer little instructive information. EPA (1980a) describes erosive soils as those with low organic matter, high amounts of silt or fine sand (e.g., loess), a blocky structure (e.g., clay), and low permeability (e.g., calichi). In contrast, the least erosive soils contain high levels of organic matter, are low in silt or fine sand, have a fine granular or crumbly structure, and are highly permeable. Thus soil type should be incorporated into the assessment of surface erosion risk.

Drainage Characteristics. The two types of soil erosion of concern in this subsection are rill or gully erosion and splash or sheet erosion. The former is more impressive and more easily observed, but the latter may be equally damaging, especially on rangelands and farmlands; however, it is more difficult to assess. It is best to prevent both types by proper land management. Where gullies exist they can be mitigated by sets of check dams and revegetation with stem-sprouting vegetation. Splash erosion is best controlled by revegetation and mulching. Because roads are vulnerable to both sheet and gully erosion, we recommend that information describing road density and mitigation be included in conservation plans (see Section 14.2.5).

Vegetative Cover and Organic Litter. Rainfall, slope, and soil texture and structure can be controlled relatively little; however, vegetation can be managed and proper management of plant cover also improves surficial soil texture and structure. For example, Packer and Laycock (1969, in Heady and Child 1994) report that plant and litter cover account for 50%–80% of the variance in erosion studies on rangelands. For the erosive granitic soils of southwest Idaho, Packer (1951, in Heady and Child 1994) recommends 70% cover with vegetation and litter where perennial grasses dominate and 90% in landscapes dominated by annual grasses. This translates to bare soil patches an average of < 10 cm in diameter in the former case and < 5 cm in the latter. Ground cover of 70% is also recommended for the sagebrush-wheatgrass assemblage of western Wyoming to reduce soil compaction and bulk density (Packer 1963, in Heady and Child 1994). On more humid grasslands, Ellison (1950, in Heady and Child 1994) reported a yield of 1.2 tons per hectare of splash erosion when there was 7 tons per hectare of herbage and litter, but 170 tons per hectare from bare soil. Clearly, increases in the amount of bare ground, soil bulk density, and devegetation—whether by over grazing, agriculture, or deforestation—produces increased runoff and soil erosion. Over sufficient time and at sufficient intensities, these uses have led to desertification in arid and semiarid environments.

14.2.3 Riparian Buffers

Riparian Functions in Relation to Buffer Width

Key Issues. Our review of the literature (Section 3.9) revealed six specific functions of riparian zones that are essential to the development and maintenance of aquatic habitats favorable to salmonids. Riparian vegetation provides shade to stream channels, contributes large woody debris to streams, adds small organic matter to streams, stabilizes streambanks, controls sediment inputs from surface erosion, and regulates nutrient and pollutant inputs to streams. In addition to these functions that directly influence aquatic habitats, riparian areas are critical habitats for a variety of terrestrial and semi-aquatic organisms and serve as migration or dispersion corridors for wildlife species (FEMAT 1993). Many of these benefits derive from the availability of water and unique microclimates in these zones. Long-term conservation of salmonids requires protecting not only the immediate functions that riparian vegetation provides, but the ecological conditions within the riparian zone needed to maintain natural vegetation communities (e.g., soil productivity, microclimate) as well. Although riparian buffers alone are insufficient to ensure healthy salmonid habitats, there is consensus in the scientific community that protection of riparian ecosystems should be central to all salmonid conservation efforts on both public and private lands (FEMAT 1993; Cederholm 1994; Cumins et al. 1994; Rhodes et al. 1994; Murphy 1995; and others).

Removal of riparian vegetation through logging, grazing, agriculture, or other means can diminish each of the important functions listed above (see review in Chapter 6). The removal of overhead cover results in more extreme temperatures during both the summer and winter through greater radiative heating and cooling. The lack of recruitment and active removal of large woody debris has left many streams in the Pacific Northwest depleted of large wood that is essential in creating pool and off-channel habitats, retaining sediments and organic materials (including salmon carcasses), creating hydraulic and physical complexity, and providing overhead cover for salmonids. The loss of root matrices of riparian trees and shrubs destabilizes streambanks, allowing banks to slough and collapse during high flow events.

Reductions in understory vegetation and disturbance to the organic litter layer permits raindrops to directly hit the soil, facilitating detachment and transport of soil to the stream channel. Alteration of riparian vegetation can also increase nutrient loadings to streams and allow chemical (e.g., pesticides, fertilizers) and biological (e.g., bacteria) contaminants associated with land-use practices to enter the stream.

Fish-bearing streams are influenced not only by the condition of adjacent riparian areas, but conditions of upstream reaches as well, including ephemeral and perennial nonfish-bearing streams. Sediments generated from unprotected upstream reaches are transported and deposited downstream, filling pools and decreasing channel stability. Removal of large trees from headwater areas may reduce recruitment of wood to downstream areas. Temperature increases caused by canopy removal in small streams can also affect downstream reaches. Because these influences of land management propagate downstream, protection of riparian zones along nonfish-bearing streams and ephemeral channels is also needed to maintain salmonid habitats.

Recommendations. We recommend that habitat conservation plans and other conservation agreements include a comprehensive plan for protecting riparian areas along all fish-bearing and nonfish-bearing streams, including ephemeral channels. Riparian buffers should be established for all land-use types and should be designed to maintain the full array of ecological processes (i.e., shading, organic debris inputs, bank stability, sediment control, and nutrient regulation) needed to create and maintain favorable conditions through time. Consideration should also be given to protecting microclimatic conditions (temperature, humidity, wind speed, soil moisture, etc.) to ensure the persistence of natural vegetation communities and, where applicable, other riparian-dependent terrestrial and semi-aquatic species.

Conservation plans should include an assessment of current and historical riparian conditions for the entire watershed (or in the case of very large watersheds, the portion reasonably affected by the HCP) with the objectives of determining the degree to which riparian functions have been altered (if at all) by past land-use practices, projecting recovery periods for various riparian functions, and identifying strategies for accelerating recovery. This analysis should include an overall assessment of cumulative effects and maps of current riparian conditions. The Federal Agencies are currently in the process of developing analytic modules that specifically address riparian functions. Washington State has developed a riparian function module that addresses current riparian conditions, long-term recruitment of large woody debris, and canopy closure/stream temperatures. The functions of nutrient cycling, litter inputs, sediment control, bank stabilization, and microclimate protection are not explicitly addressed in the riparian modules of either the Federal or State of Washington guides.

Evaluation Criteria. The establishment of riparian buffer zones is generally accepted as the most effective way of protecting aquatic and riparian habitats (Cumins et al. 1994). We define buffer zones as areas adjacent to the stream channel or floodplain in which land-use activities are prohibited or substantially restricted. In most instances, riparian management can be divided into two components: delineation of appropriate riparian buffer widths and determination of allowable activities within the riparian buffer zone. Both of these components can be addressed by considering the functional roles of the riparian zone, and particularly those of riparian vegetation.

A functional approach to riparian protection requires a consistent definition of riparian ecosystems based on "zones of influence" for specific riparian processes. In constrained reaches, the active stream channel remains relatively stable through time and riparian zones of influence may be defined based on site-potential tree heights and distance from the active channel. In unconstrained reaches with braided or shifting channels and broad floodplains, the riparian area of influence is more difficult to define. In these reaches, it is more appropriate to define the riparian zone based on the extent of the floodplain, rather than the active channel, because movement of the active channel across the floodplain through time may render buffer strips ineffective. Consequently, it is reasonable to propose buffers of varying absolute widths based on specific reach-level characteristics. Riparian Reserves for Federal lands (FEMAT 1993; FS and BLM 1994c) incorporate these ideas by defining riparian reserves based on multiple criteria. For example, the boundaries for Riparian Reserves surrounding fish-bearing streams are defined by five potential criteria: 91 m (300 ft) slope distance on each side of the channel, two site-potential trees, the outer edges of the 100 year floodplain, the distance from the active channel to the top of the inner gorge, or to the outer edges of riparian vegetation, whichever is greatest.

The effectiveness of riparian buffers can be best evaluated within the context of specific protection goals. For example, riparian standards designed to protect only salmonid habitats would differ substantially from standards to protect other riparian-dependent species, including amphibians, birds, mammals, and reptiles. Consequently, it is

reasonable to expect more conservative riparian protection strategies for a multi-species HCP than for one designed for protecting only salmonids. In the sections below, we review literature pertaining to the buffer widths required to provide full protection to specific riparian functions identified as critical in the technical foundation (Section 3.9). For some functions, these relationships are not entirely clear and these uncertainties are noted.

Stream Shading. The ability of riparian forests to provide shade to stream chaiinels is a function of numerous site-specific factors including vegetation composition, stand height, stand density, latitude (which determines solar angle), topography, and orientation of the stream channel. These factors influence how much incident solar radiation reaches the forest canopy and what fraction passes through to the water surface. The shading influence of an individual tree can be expressed geometrically as a function of tree height, slope, and solar angle. For example, Broderson (1973) notes that in mid-July at 45°N latitude, a 61 m (200 ft) high tree on level terrain provides shade 27 m from its base. The same tree provides shade a slope distance of 41.6 m from its base (i.e., 36.6 m measured horizontally from the stream edge) on a 31° slope and 68.8 m (48.9 m horizontal) from its base on a 45" slope. These values represent the *maximum* potential zone of influence for a tree of this height at this latitude and time of year. In natural forests, stand density and composition may moderate the shading influence of trees within this zone with trees closer to the stream channel and understory shrubs providing the majority of stream shade.

The most thorough studies of the effectiveness of riparian buffer strips have been conducted in the Cascade and Coast Ranges of western Oregon. Brazier and Brown (1973) found that angular canopy densities comparable to old-growth stands (i.e., 80%–90%)could be attained with buffers of approximately 22–30 m for coniferous forests in the southern Cascades and Oregon Coast Range. Data from Steinblums et al. (1984) suggests buffers greater than 38 m are needed to retain 100% of natural shading in coniferous forests of the western Cascades (610–1220 m elevation). Based largely on these data, several authors have coiicluded that buffers of 30 m or more provide adequate shade to stream systems (Murphy 1995; Johnson and Ryba 1992; Beschta et al. 1987). The generalized curves presented by FEMAT (1993) suggest that cumulative effectiveness for shading approaches 100% at a distance of approximately 0.75 tree heights from the stream channel (see Figure 3-2). This translates to 25.1 m and 38.9 m for forests with average tree heights of 33.5 m (110 ft) and 51.8 m (170 ft), respectively.

The apparent consensus that buffers exceeding 30 in are needed for stream shading has been based largely on studies in the Cascade and Coast Ranges. There is little published information regarding buffer widths needed to provide natural levels of shade for streams in eastside forest, rangeland, and agricultural systems. Eastside forests, particularly old-growth ponderosa pine forests, have lower stem densities and crown-closure than westside Douglas-fir-dominated systems and frequently lack the dense understory vegetation typical of many westside riparian areas. Consequently, the width of buffers needed to maintain full shading may differ. For hardwood-dominated riparian forests that were once common along streams east of the Cascades, appropriate buffer widths for shade are even less certain, in part because examples of intact riparian ecosystems are extremely rare. More research on riparian influences on shading for all ecosystems east of the Cascades is needed before specific criteria can be recommended; however, in most instances, buffer widths designed to protect other riparian functions (e.g., LWD recruitment) are likely to be adequate to protect stream shading.

LWD Recruitment. Large wood enters stream channels by a variety of mechanisms, including toppling of dead trees, windthrow, debris avalanches, deep-seated mass soil movements, undercutting of streanibanks, and redistribution from upstream (Swanson and Lienkamper 1978). Most assessments of buffer widths required for maintaining natural levels of large wood have considered only wood delivered by toppling, windthrow, and bank undercutting. Yet in some systems, wood delivered from upslope areas (via mass wasting) or upstream reaches (via floods or debris torrents) may constitute a significant fraction of the total wood present in a stream reach. In attempting to identify sources of large wood pieces in 39 stream reaches, McDade et al. (1990) failed to account for more than 47% of the woody debris pieces, suggesting that upslope and upstream sources potentially may be quite important. These mechanisms of delivery are more difficult to model, thus the discussion below focuses on recruitment from the immediate riparian zone. Nevertheless, in evaluating habitat conservation plans, consideration should be given to potential recruitment of wood from upslope areas and nonfish-bearing channels.

The potential for a tree or portions of a tree to enter the stream channel by toppling, windthrow, or undercutting is primarily a function of slope distance from the stream channel in relation to tree height and slope angle. Consequently, the zone of influence for

large wood recruitment is defined by the particular stand characteristics rather than an absolute distance from the stream channel or floodplain. Other factors, including slope and prevailing wind direction, niay influence the proportion of trees that fall in the direction of the stream channel (Steinblums et al. 1984; Robison and Beschta 1990b; McDade et al. 1990); however, if the goal is to maintain full recruitment of large wood to the channel, then protection of all trees within the zone of influence is desirable.

FEMAT (1993) concluded that the probability of wood entering the active stream channel from greater than one tree height is generally low (see Figure 3-2). Exceptions occur in alluvial valleys, where stream channels may shift in response to sediment deposition and high flow events. Two models of large wood recruitment also assume that large wood from outside of one tree height seldom reaches the stream channel (Van Sickle and Gregory 1990; Robison and Beschta 1990). Murphy and Koski (1989) found that 99% of all identified sources of LWD were within 30 m of the stream channel in hemlock and Sitka spruce forests of southeastern Alaska with site potential tree heights of approximately 40 m (131 ft) (M. Murphy, personal communication). Their study defined LWD as pieces greater than 3 m length and 10 cm diameter and thus excluded smaller fractions classified as large wood in other studies. In addition, because trees far from the stream channel generally contribute smaller individual pieces (i.e., the tops of trees) that are more easily transported downstream, the authors' abilities to identify sources likely decreased with increasing distance from the channel. Consequently, protecting all LWD recruitment may require slightly larger buffer zones. McDade et al. (1990) examined LWD recruitment to streams at 37 sites in the Cascade and Coast Ranges of Oregon and Washington and found that source distances were as far as 55 m in old-growth (> 200 years) coniferous forests and 50 m in unmanaged, mature (80–200 year old) conifer stands. Tree heights averaged 57.6 m in old-growth stands and 48 m in mature stands; thus, source distances were approximately equal to one site-potential tree height. In this study, woody debris was defined as pieces greater than 1 m length and 0.1 m diameter at the small end. Cederholm (1994) reviewed the literature regarding recommendations of buffer widths for maintaining recruitment of LWD to streams and found most authors recommended buffers of 30–60 m for maintaining this function. In summary, most recent studies suggest buffers approaching one site-potential tree height are needed to maintain natural levels of recruitment of LWD.

An additional consideration in determining appropriate activities in riparian zones relative *to* large wood recruitment is the potential size distribution of LWD. Murphy (1995) notes that larger pieces of wood form key structural elements in streams, serving to retain smaller debris that would otherwise be transported downstream during high flow events. Bisson et al. (1987) suggest that the size of these key pieces is approximately 30 cm or more in diameter and 5 m in length for streams less than 5 m in width and 60 cm or more in diameter and 12 m in length for streams greater than 20 m in width.

For making Endangered Species Act determinations of effect, NMFS (1985c) uses large-size fractions of wood to define properly function habitats. These key pieces are defined as greater than 60 cm in diameter and 15 m in length for westside systems and greater than 30 cm in diameter and 11 m in length for eastside systems. Consequently, riparian protection plans need to ensure not only an appropriate amount or total volume of wood, but pieces of sufficient size to serve as "key pieces" (Murphy 1995).

Fine Organic Litter. Smaller pieces of organic litter (leaves, needles, branches, tree tops, and other wood) enter the stream primarily by direct leaf or debris fall, although organic material may also enter the stream channel by overland flow of water, mass soil movements, or shifting of stream channels in unconstrained reaches. Little research has been done relating litter contributions to streams as a function of distance from the stream channel; however, it is assumed that most fine organic litter originates within 30 m, or approximately 0.5 tree heights from the channel (FEMAT 1993). In deciduous woodlands, windborne leaf litter may travel farther from source trees than needles or twigs from coniferous vegetation; consequently, riparian buffers may need to be wider than suggested above to protect natural levels of organic inputs. Nevertheless, in most cases buffers designed to protect 100% of LWD recruitment will likely provide close to 100% of small organic litter as well.

Bank Stabilization. Roots of riparian vegetation help to bind soil particles together, making streambanks less susceptible *to* erosion. In addition, riparian vegetation provides hydraulic roughness elements that dissipate stream energy during high or overbank flows, further reducing bank erosion. In most instances, vegetation immediately adjacent to the stream channel is most important in maintaining bank integrity (FEMAT 1993); however, in wide valleys with shifting stream channels, vegetation throughout the floodplain may be important over longer time periods, Although data quantifying the effective zone of influence relative to root strength is scarce, FEMAT (1993) concluded that most of the

stabilizing influence of riparian root structure is probably provided by trees within 0.5 potential tree heights of the stream channel. Consequently, buffer widths for protecting other riparian functions (e.g., LWD recruitment, shading) are likely adequate to maintain bank stability. In addition, consideration should be given to the composition of riparian species within the area of influence because of differences in the root morphology of conifers, deciduous trees, and shrubs varies. Specific relationships between root types and bank stabilization have not been documented; however, if the purpose of riparian protection is to restore natural bank characteristics, then retaining natural species composition is a reasonable target for maintaining bank stabilization function of riparian vegetation.

Sediment Control. The ability of riparian buffers to control sediment inputs from surface erosion depends on several site characteristics including the presence of vegetation or organic litter, slope, soil type, and drainage characteristics. These factors influence the ability of buffers to trap sediments by determining the infiltration rate of water and the velocity (and hence the erosive energy) of overland flow. Several recent reviews have examined the relationship between buffer width and sediment retention (Belt et al. 1992; Castelle et al. 1992; Johnson and Ryba 1992) and tlie information below is taken primarily from these sources.

Several studies have examined effectiveness of buffers in controlling sediments from forested lands. Broderson (1973) concluded that buffer widths of 15 ni controlled most sediments on slopes less than 50% (26°) and that buffers of 61 m were effective on extremely steep slopes in watersheds of western Washington. Corbett and Lynch (1985) recommended buffers of 20–30 m for controlling sediments. In Pennsylvania, Lynch et al. (1985) concluded that buffers of 30 m removed 75%–80% of suspended sediments in stormwater draining areas that had been clearcut and burned, but that greater sedimentation occurred in areas that were logged and subsequently treated with herbicides. FEMAT (1993), citing these same studies, concluded that buffers of approximately one site potential tree were probably adequate to control sediments from overland flow.

Belt et al. 1992 provide a thorough review of studies examining sediment transport below roaded areas on forested soils and drew four conclusions related to riparian buffers strip design: 1) riparian buffers should be greater where slopes are steep, 2) riparian buffers are ineffective in controlling sediments resulting from channelized flows that originate outside of the riparian buffer, 3) sediments rarely travel more than 91 m, unless flows are channelized, and 4) removal of natural obstructions to flow—vegetation, woody debris, rocks, etc.—within the buffer increases the travel distance of sediments, Johnson and Ryba (1992) reviewed three studies of buffer effectiveness in reducing sediments in runoff from agricultural lands and found recommendations ranging from 3 m for sandy soils up to 122 m for clay soils (Wilson 1967). Gilliam and Skaggs (1988) reported 50% deposition of sediments within the first 88 m of a vegetated buffer adjacent to agricultural fields. Recommendations of the Soil Conservation Service (SCS 1982) call for buffers in agricultural lands of 8–46 m depending upon slope. Belt et al. (1992) concluded that while studies support tlie use of buffer strips as a means of filtering sediment from agricultural lands, they provide no definitive means for determining appropriate buffer widths.

Because of the high degree of variability in the effectiveness of buffers, we cannot draw any definitive conclusions regarding buffer widths required for sediment control. On gentle slopes, buffers of 30 m may be sufficient to filter sediments, whereas on steeper slopes, buffers of 90 m or more may be needed. In addition, riparian buffers are most effective in controlling sediments from sheet erosion and have less influence on sediments that reach the stream channel via channelized flow (Broderson 1973; O'Laughlin and Belt 1994; Murphy 1995), although Megahan and Ketcheson (1996) reported that obstructions (logs, trees, and rocks) significantly reduced the travel distance of granitic sediments in concentrated flows below forest roads on Idaho. We suggest that, except on steep slopes, buffers designed to protect other riparian functions will generally control sediments to the degree that they can be controlled by riparian vegetation. It is essential, however, that riparian protection be compleniented with practices for minimizing sediment contributions from outside the riparian area, particularly those from roads and associated drainage structures, where large quantities of sediment are often produced. In addition, activities within the riparian zone that disturb or compact soils, destroy organic litter, remove large down wood, or otherwise reduce the effectiveness of riparian buffers as sediment filters should be avoided.

Nutrients and Other Dissolved Materials. Riparian vegetation takes up nutrients and other dissolved materials as they are transported through the riparian zone by surface or near-surface water movement. However, the relationship between buffer width and filtering capacity is less well understood than other riparian functions. Those studies that have been published indicate substantial variability in the effectiveness of buffer strips in controlling nutrient inputs (reviewed in Castelle 1992; Johnson and Ryba

1992). The required buffer width for filtering nutrients and other dissolved materials depends on the specific type and intensity of land use, type of vegetation, quantity of organic litter, infiltration rate of soils, slopes, and other site-specific characteristics.

Lynch et al. (1985) observed significant increases in levels of nitrate-nitrogen following logging of a mixed-deciduous forest in Pennsylvania where 30 m buffers were retained; however, they concluded that these levels were not detrimental to stream biota. In the northern Rockies, increases in numerous chemical parameters (pH, bicarbonate, nitrate, sulfate, potassium, calcium, and magnesium) were recorded in surface waters adjacent to three areas that were clearcut and burned, but where undisturbed buffers measuring 30–61 m were retained (Snyder et al. 1975). These results suggest that even fairly wide buffer strips may not prevent elevation of some chemicals following logging, particularly if water is routed to the stream via channelized flow, rather than overland flow through the riparian buffer strip.

Several studies have examined the potential effectiveness of vegetated filter strips in retaining nutrients from agricultural systems. Dillaha et al. (1989) reported that 9.1 m vegetated filter strips removed 79% of phosphorous and 73% of nitrogen on experimental plots in Virginia. Madison et al. (1992; cited in Castelle 1994) reported that 9.1 m grass buffer strips removed approximately 96.0%–99.9% of phosphate phosphorous, nitrate-nitrogen, and ammonium-nitrogen. Xu et al. (1992; cited in Castelle 1994) reported greater than 99% reduction in nitrate-nitrogen in soils in a 10 m mixed herbaceous and forested buffer strip in North Carolina. Nutrient and bacteria levels in runoff from poultry and dairy farms or direct manure applicatioiis may be substantially higher than from other agricultural lands; consequently, buffers may need to be wider. Vanderholm and Dickey (1978) monitored natural runoff from feedlots and found that buffer widths of 91 m on a 0.5% slope and 262 m on a 4.0% slope removed 80% of nutrients, suspended solids, and oxygen demanding substances from surface runoff (cited in Johnson and Ryba 1992). Shisler et al. (1987) reported that wooded riparian buffers in Maryland removed 89% of excess nitrogen and 80% of excess phosphorous from animal wastes with most of the removal being achieved within 19 m. Doyle et al. (1977) found that forest and grass buffer strips of approximately 4 m reduced nitrogen, phosphorous, potassium and fecal bacteria levels in runoff from manure applications, but they did not indicate the percent reduction in these materials. Young et al. (1980; cited in Johnson and Ryba 1992) recommended buffer widths of 36 m for controlling nutrients in runoff from feedlots. Two studies have proposed that buffer strip width should be a function of the total area affected by animal wastes. A 1:1 buffer area to waste area ratio has been suggested as sufficient to reduce nutrients from poultry manure to background levels (Bingham et al. 1980). Similarly, Overcash et al. 1981 reported that a 1:1 buffer area to waste area reduced animal waste concentrations by 90%–100%.

Little information exists regarding the effectiveness of buffers in filtering runoff in urban areas. One exception is the study of Phillips (1989), who modeled pollutant removal efficiencies from residential areas. He found that buffers of 22.9 m required for estuarine shorelines in North Carolina were inadequate for reducing nitrogen, phosphorous, and BOD of runoff from residential areas.

Because of the variability observed in the effectiveness of buffers in controlling input of nutrients and other dissolved materials, it is difficult to recommend specific criteria for buffer widths for this function. Belt et al. (1992) concluded that although the utility of buffer strips in reducing nutrient loading has been demonstrated for forested and agricultural systems, existing studies do not provide an adequate basis for determining effective buffer widths. The studies of Snyder et al. (1975) and Lynch et al. (1985) cited above indicate that nutrient increases from logging and burning may occur even with fairly large buffers (30–60 m), however, these nutrient increases represent whole-watershed responses to logging and larger buffers may have little value in further reducing nutrient loads. Based on the above review, we suggest that for most forest lands, buffers designed to protect other riparian functions (e.g., LWD recruitment, shading) are probably adequate for controlling nutrient inputs to the degree that such increases can be controlled by buffers. Exceptions may occur when fertilizer or other chemical applications result in high concentrations of nutrients in surface runoff.

For rangelands, agricultural systems, and urban areas, we believe current understanding is insufficient to make specific buffer recommendations. The review of Johnson and Ryba (1992) suggests that effective buffers for nutrient control on forest and grasslands range from approximately 4–42 m, but that substantially wider buffers are needed to control nutrients and bacteria (fecal coliform) from feedlot runoff. We recommend that buffer widths for nutrient and pollution control on these lands be tailored to specific site conditions, including slope, degree of soil compaction, vegetation characteristics, and intensity of land use. In many instances, buffer widths designed to protect LWD recruitment and shading may be adequate to prevent excessive nutrient or pollution concentrations. However, where land use activity is especially intense, buffers for

protecting nutrient and pollutant inputs may need to be wider than those designed to protect other riparian functions, particularly when land-use activities may exacerbate existing water quality problems. Buffers need to be accompanied by other protective measures when drainage structures (e.g., irrigation canals, drain tiles) bypass the riparian zone.

Riparian Microclimate and Productivity. Changes in microclimatic conditions within the riparian zone resulting from removal of adjacent vegetation can influence a variety of ecological processes that may affect the long-term integrity of riparian ecosystems. However, the relationship between buffer width and riparian microclimate has not been documented in the literature. FEMAT (1993) presented generalized curves relating protection of microclimatic variables relative to distance from stand edges into forests (see Figure 3-3). These curves suggest that buffers need to be extended an additional one-to-two tree heights outside of the riparian zone to maintain natural levels of soil moisture, solar radiation, and soil temperature within the riparian zone and even larger buffers (up to three tree heights) to maintain natural air temperature, wind speeds, and humidity. The recommendations of FEMAT (1993) were based on studies in upland forests in the Cascades (Chen 1991), and their applicability to riparian zones is uncertain (O'Laughlin and Belt 1994). Therefore, additional research is needed before we can confidently suggest buffer widths that are likely to protect riparian microclimate.

The long-term productivity of riparian habitats may also be affected by management in adjacent upland forests. Decaying logs in the riparian zone may be important sites for germination of many types of vegetation because they retain moisture and tend to shed leaf litter that can bury seedlings (reviewed in Harmon et al. 1986). In particular, rotting logs in forests of western hemlock and sitka spruce appear to be key sites for germination. McKee et al. (1982) found that 94%–98% of all seedlings in forests of hemlock and spruce on the Olympic Peninsula were growing on LWD that constituted only 6%–11% of the forest floor. Christy and Mack (1984) found that 98% of all western hemlock seedlings were associated with rotting large wood in a mixed old-growth forest of hemlock and Douglas-fir. In this study, only 6% of the total forest floor area was covered with LWD. Harmon et al. (1986) urge some caution in interpreting these results because the relationship between seedling establishment and long-term survival is not known.

Large wood is also an important source of nutrients and organic matter in riparian forests. In an old-growth, Douglas-fir forest in the western Cascades, Sollins et al. (1980; cited in Maser et al. 1988) found that stems of fallen trees contained 46%, 30%, and 12% of the total nitrogen, potassium, and phosphorous, respectively, found on the forest floor. Means et al. (unpublished data; cited in Maser et al. 1988) found that about 30% of all soil organic matter in two old-growth Douglas-fir forests was contained in downed trees of 500 years age or older. Sollins et al. (1980) found that proportion of soil organic matter from LWD was four-fold higher than in other forms of forest litter. These studies suggest that long-term integrity of riparian areas may be dependent on adequate recruitment of large wood to the forest floor from within the riparian zone and adjacent uplands. They also suggest that the practice of removing down logs from within the riparian zone and placing them in channels may affect long-term riparian productivity. Maintaining recruitment of wood to the riparian zone (not just the stream channel) would require extending buffer zones beyond the edge of the defined riparian zone.

Wildlife Habitat. The importance of riparian areas to many wildlife species is well documented (see review Section 3.9.8). However, generic recommendations for riparian buffers to protect wildlife are not justifiable because each species has unique habitat requirements. Johnson and Ryba (1992) reviewed the literature related to buffer widths for wildlife protection and found recommended buffer widths to be highly variable, ranging from 10–200 m. Suggested buffer widths by taxa included: 30–100 m for beaver, 67–93 m for small mammals, 100 m for large mammals, and 75–200 m for birds. Requirements for amphibians and reptiles were not included in their review; however, most amphibians require cool, moist habitats throughout their life cycles and many species are commonly found associated with large woody debris (FEMAT 1993). Consequently, maintaining microclimatic conditions and recruitment of LWD within the riparian zone may be essential for protecting amphibians. FEMAT (1993) also conducted a review and found studies recommending buffers from 30–183 m in width for wildlife protection; they did not, however, base riparian reserve widths on wildlife needs. Cumins et al. (1994) also noted the importance of riparian zones for wildlife but did not incorporate wildlife needs into their buffer width recommendations.

Buffers and Windthrow. Trees within riparian buffers that are immediately adjacent to clearcuts have a greater tendency to topple during windstorms than trees in undisturbed forests. Extensive blowdown can potentially affect aquatic ecosystems in a number of ways, both positive and negative. In stream systems that lack wood because of past management practices, blowdown may immediately

benefit salmonids by providing structure to the channel. Over the long-term, however, blowdown of smaller trees may hinder the recruitment of large wood pieces that are key to maintaining channel stability and that provide habitats for vegetation and wildlife within the riparian zone. In addition, soil exposed at the root wads of fallen trees may be transported to the stream channel, increasing sedimentation. Other riparian functions, including shading, bank stabilization, and maintenance of riparian microclimates may also be affected. Rhodes et al. (1994) suggest that buffers need to extend to a distance of two site-potential tree heights (or > 91 m) to protect riparian buffers from windthrow; however, local site conditions dictate vulnerability of stands to windthrow and appropriate buffer widths would vary accordingly.

Effectiveness of Federal and State Forest Practices in Maintaining Riparian Functions

The review in the preceding section provides a framework for assessing the relative protection afforded specific riparian processes by riparian management guidelines currently in effect on Federal and nonfederal lands. Riparian management guidelines have been most completely developed for forested lands on both public and private lands. Riparian management guidelines for Federal lands within the range of the northern spotted owl are detailed in the Record of Decision (ROD) for the President's Forest Plan (FS and BLM 1994c); these guidelines apply to much of the region in western Oregon, Washington, and California. Interim riparian protection measures for managing anadromous fish-producing waters on Federal lands outside the range of the northern spotted owl (i.e., eastern Washington, Oregon, and California, and all salmon-bearing steams in Idaho) are contained in PACFISH (FS and BLM 1994a). Interim riparian protection measures for streams with resident (nonanadronious) native fish in eastern Oregon, eastern Washington, Idaho, western Montana, and portions of Nevada are detailed in INFISH (FS 1995). Forest practices in riparian areas on nonfederal lands are regulated by forest practices rules specific to each State. At present, no coniparable protections exist for range, agricultural, and urban lands.

ROD, PACFISH, INFISH, and the States each define the width of riparian management zones and allowable activities within the riparian zone based on water-type classifications. Streams on Federal lands are classified based on presence or absence of fish, whether the stream is intermittent, and whether the stream is in a key or nonkey watershed (Table 14-1). Ponds, lakes, and wetlands are classified based on size and whether they are natural or constructed. Water classification systems for Idaho, Washington, Oregon, and California are more variable. All of these States use presence or absence of fish to classify streams, but additional classification variables are used, including other beneficial uses (e.g., domestic water supplies: all States) stream width (ID, WA), mean annual flow (OR), substrate type (WA), bank side-slope angle (CA), and whether the stream is capable of downstream sediment transport (CA) (Table 14-1).

Federal riparian reserves (ROD) or riparian habitat conservation areas (PACFISH, INFISH) differ from the riparian management zones of the States both in terms of how riparian zone widths are defined and the level of activity allowed within the riparian zone. ROD, PACFISH, and INFISH define riparian reserve widths based on site-potential tree heights, whereas all of the State forest practice rules have fixed-width riparian management zones, though in some states these widths may be increased on steep slopes or highly erodible soils (ID, CA, WA). The Federal strategies allow timber harvest and other activities within riparian reserves (ROD) or riparian habitat conservation areas (PACFISH, INFISH) only if such activities will not compromise Aquatic Conservation Strategy (ROD) or Riparian Management (PACFISH, INFISH) objectives or if such activities are needed to attain these objectives. In contrast, all four States generally allow greater activity within the riparian management zone. State forest practice rules seek to protect riparian shading and LWD recruitment through retention of 1) a percentage of overstory and understory vegetation (all States), 2) a specified basal area of conifers per length of stream or per acre (OR), **3)** a specified number of trees per length of stream (ID, CA, WA, OR), or 4) a specified number of trees of various dimensions per length of stream (ID). The width of the riparian zone and the degree of human disturbance allowed within the zone for each stream class varies by State (Table 14-1). In addition, some States have different buffer widths or leave tree requirements depending on the district or region (WA, OR) or the type of harvest (clearcut vs. partial cut or thinning, OR) or yarding method (CA). In Washington State, watershed analysis can be used to justify smaller or larger buffers and more or less harvest within riparian zones as long as riparian functions are not impaired. Similarly, California allows increases or decreases in riparian management zone width and canopy retention requirements based on site characteristics or proposed forestry practices, provided they do not degrade beneficial uses. These changes must be approved by a Registered Professional Forester or the Director of Forestry and Fire Protection.

Because of the different classification schemes and inconsistent leave-tree requirements, it is not possible to quantitatively compare the effectiveness of the State forest practice rules in protecting riparian

Table 14-1. Riparian management regulations for Federal, State, and private forest lands in Idaho, Oregon, Washington, and California. SPT = site potential tree, SPZ = stream protection zone, RMA = riparian management area, WLPZ = water course and lake protection zone, RR = riparian reserve, RHCA = riparian habitat conservation area. Modified from Stephen Phillips, Pacific States Marine Fisheries Commission.

Regulating Authority	REQUIREMENTS					
	STREAM SHADE	TREE RETENTION				
IDAHO Idaho Code Title 38, Chap. 13 Rules 1-8	*Class I Streams* Leave 75% of current stream shade.		Stream Class/Width			
			Class IA	Class IB	Class IC	Class II [1]
			Over 20' fish-bearing [2]	10'–20' fish-bearing [2]	Under 10' fish-bearing [2]	headwater, few if any fish
		Tree Diameter (DBH)	Minimum Number of Standing Trees per 1000' (each side)			
		0–7.9"	200	200	200	--
		8–11.9"	42	42	42	--
		12–19.9"	21	21	--	--
		20+"	4	--	--	--
		SPZ Width [3,4]	75'	75'	75'	5'
	Class II Streams No specific shade requirements Leave undisturbed soils to a minimum of 5' width.					
OREGON OAR 629-24 OAR 629-57	For all fish-bearing and and large and medium nonfish-bearing streams retain all understory vegetation within 10 A of the high water level and all trees within 20 ft of the high water level		Stream Type/Width/Flow			
			Type F1	Type FII	Type FIII	Type N-D [5]
			Large, fish-bearing >25' width	Medium, fish-bearing 8'–25' width	Small, fish-bearing <8' width	Small, non-fish-bearing <8' width
			>10 cfs avg. annual flow	2–10 cfs avg. annual flow	<2 cfs avg. annual flow	<2 cfs avg. annual flow
		Tree Retention [6]	Square Feet of Conifer Basal Area per 1,000' of Stream (each side) [7]			
		Coast Range	170–230	90–120	20–40	n.a.
		W. Cascades/Interior	200–270	110–140	20–40	n.a.
		Siskiyou	170–220	90–110	20–40	n.a.
		Eastside	130–170	70–90	50 [8]	n.a.
		RMA Width	100'	70'	50'	0–70'
WASHINGTON WAC 222-30 WAC 222-16-30	Applies only if there is harvest planned within RMZ. once stream temperature and elevation are determined, a nomograph is used to derive minimum canopy cover Additional requirements if >25% of canopy removed and temperature model indicates stream temperature increase >2 8°C		Stream Type/Width			
			Type 1 & 2 Water	Type 1 & 2 Water	Type 3 Water	Type 4 Water
			>75' width	<75' width	<5' width	>2' width
			State shorelines or substantial fish numbers [9]	State shorelines or substantial fish numbers [9]	Significant fish numbers [9]	Nonfish-bearing perennial or intermittent
		Substrate Type	Trees per 1,000' of stream (westside)/Trees >4"DBH per acre (eastside) [10]			
		Gravel/Cobble	50 / 135	100 / 135	25 / 135	When needed to protect resource
		Boulder/Bedrock	25 / 75	50 / 75	25 / 75	
		RMZ Width [11] West	25'-100'	25'-75'	25'	25'
		RMZ Width [11] East	50' avg.	50' avg.	50' avg.	--
CALIFORNIA 14 CCR 916 (936, 956)	*Class I Streams* Retain at least 50% overstory & understory canopy covering ground ground and 25% overstory conifers		Stream Class [12]			
				Class I [13]	Class II [14]	Class III
				Fish always or seasonally present; fish spawning or migration habitat	Fish always or seasonally present within 1,000 feet downstream	Nonfish-bearing capable of sediment transport
	Class II Streams Retain 50% total canopy covering ground, retain 25% of overstory conifers. *Class III Streams* 50% of total understoty vegetation shall be left.		Side Slope Class	Minimum no. conifers >16" DBH & >50' height per acre		
				≥2	≥2	--
		WLPZ Width [15]	<30%	75'	50'	Determined on site-specific basis
			30%–50%	100'	75'	
			>50%	150'	100'	

Table 14-1. Riparian management regulations for Federal, State, and private forest lands in Idaho, Oregon, Washington, and California. SPT = site potential tree, SPZ = stream protection zone, RMA = riparian management area, WLPZ = water course and lake protection zone, RR = riparian reserve, RHCA = riparian habitat conservation area. Modified from Stephen Phillips, Pacific States Marine Fisheries Commission.

Regulating Authority	REQUIREMENTS: STREAM SHADE	TREE RETENTION				
ROD (FS & BLM 1994c)	No specific shade requirements; shade protected through large riparian reserves. No harvest permitted in riparian reserves when riparian conservation objectives are adversely affected, as determined by watershed analysis.		Stream or Lake Type [16]			
			Type I	Type II	Type III	Type IV
		Western OR, WA, and CA within range of northern spotted owl.	Fish-bearing	Permanent nonfish-bearing	Lakes/natural ponds	Intermittent streams, wetlands <1 ac.
		RR Width [17]	300'/2 SPT	150'/1 SPT	300'/2 SPT	100'/<1
PACFISH Alternative 4 (FS & BLM 1994a)	No specific shade requirements; shade protection through riparian reserves. No harvest in RHCAs except if natural disturbance and riparian conservation objectives not compromised.	East. OR, east. WA, ID, and portions of CA w/anadromous salmonids outside range of spotted owl.	Stream or Lake Type			
			Type I	Type II	Type III	Type IV
			Permanent, fish-bearing	Permanent, nonfish-bearing	Lakes, ponds, reservoirs, wetlands >1 ac.	Intermittent stream, wetlands <1 ac.
		RHCA Width: key	300'/2 SPT	150'/1 SPT	150'/1 SPT	100'/1 SPT
		RHCA Width: non-key	300'/2 SPT	150'/1 SPT	150'/1 SPT	50'/0.5 SPT
INFISH (FS 1995)	No specific shade requirements; shade protected through large riparian reserves. No harvest permitted in riparian reserves when riparian conservation objectives are adversely affected, as determined by watershed analysis.	East. OR, east. WA, ID, west. MT, and portion of NV with resident native fish outside range of anadromous fish.	Stream or Lake Type			
			Type I	Type II	Type III	Type IV
			Fish-bearing	Permanent nonfish-bearing	Lakes, ponds, reservoirs, wetlands >1 ac.	Intermittent streams, wetlands <1 ac.
		RHCA Width: priority	300'/2 SPT	150'/1 SPT	150'/1 SPT	100'/1 SPT
		RHCA Width: non-priority	300'/2 SPT	150'/1 SPT	150'/1 SPT	50'/0.5 SPT

[1] Idaho is currently considering changing Class II SPZ width to 30', with tree retention of 140 trees in 0-7.9' diameter class per 1,000 of stream.
[2] ID class I streams also includes those used for domestic water supplies.
[3] Widths for ID, OR, CA, ROD, PACFISH, and INFISH are based on slope distances. WA uses horizontal widths. All widths apply to each side of stream channel.
[4] ID may require wider SPZ when stream is adjacent to steep slopes or erodible soils.
[5] OR distinguishes between large, medium, and small streams for N and D type streams, Data shown above are for small streams.
[6] Tree retention requirements shown are for clearcut harvest in areas with good conifer stocking; basal area requirements are higher for partial cuts and thinning. In addition to basal area requirements, OR also has leave tree requirements of 40 live conifers per 1,000 ft of stream for large Type F streams and 30 live conifers per 1,000 ft of stream for medium Type F streams.
[7] Higher value represents "standard management target" for basal area; lower value represents "active management target." Operators may place conifer logs or downed trees in Type F streams and receive basal area credit toward meeting tree retention requirements as long as active management target is achieved.
[8] In eastside systems, snags, dead and dying trees, and hardwoods may be counted towards basal area requirements.
[9] "Substantial" and "Significant" for spawning, rearing, or migration are not defined.
[10] WA expresses leave tree requirements as trees per 1,000 ft of stream for westside systems and trees ≥4" dbh for eastside systems. For westside streams, ratio of conifer to deciduous RMZ leave trees: Type 1 & 2 waters, representative of stand; Type 3 waters <5' stream width, 1 to 1 with 6 minimum size or next largest available. For eastside streams, operator must leave all conifers ≤12" dbh plus 16 live conifers per acre from 12-20 dbh and 3 live conifers per acre > 20".
[11] RMZ width for eastside streams must average 50', with 30' minimum and 300' maximum for clearcuts. For partial cuts, minimum and maximum RMZ widths are 30' and 50', respectively.
[12] CA also delineates class IV streams: man-made water courses, usually downstream, established domestic, agricultural, hydroelectric supply, or other beneficial uses.
[13] CA class I streams also include: domestic water supplies on site and/or within 100' downstream of operation
[14] CA class II streams include those providing aquatic habitat for nonfish aquatic species.
[15] Values represent general WLPZ widths. Widths may be decreased (to a minimum of 50 ft for class I and II streams) or increased based on soil, slopes, geology, hydrology, and proposed management practices with approval from the Registered Professional Forester and the Director, provided downstream beneficial uses are not degraded.
[16] ROD has an additional class "constructed ponds, reservoirs, and wetlands" not included in this table.
[17] Riparian Reserve or RHCA width may be extended to top of inner gorge, outer edge of 100-yr floodplain, or outer edge of riparian vegetation if these distances are greater than prescribed slope distance or SPT height.

functions. Nevertheless, a qualitative sense of the level of protection afforded to specific processes can be gained based on riparian buffer width and the allowable level of activity. Figures 14-2 and 14-3 illustrate the differing buffer widths and protection levels for each class of water on Federal and nonfederal lands for eastside and westside systems. To facilitate comparison between Federal and State regulations, we converted fixed buffer widths into site-potential tree heights. We assumed a site-potential tree height of 170 feet for westside forests (Figure 14-2) and 110 feet for eastside forests (Figure 14-3) based on FEMAT (1993) designations. However, forests in the Olympics of Washington, the Siuslaw National Forest of Oregon, and the redwood zone of California contain site-potential trees in excess of 200 ft; consequently, the riparian zones of influence extend farther from the stream channel in these systems. Below we evaluate Federal and State riparian zones for Oregon, California, Idaho and Washington in terms of the protection they provide to shading, LWD recruitment, organic litter inputs, bank stability, sediment control, and nutrient control. Riparian buffer widths required to maintain 100% of each function are shown on the top of each figure and were based on the review in the preceding section. Assessing the degree of protection based on site-potential tree heights poses some difficulties. For certain functions (LWD recruitment, shading, organic litter inputs), site potential tree height is the best yardstick for assessing protection because tree height directly influences these functions. However, for other functions (*e.g.*, bank stabilization and perhaps sediment control and nutrient regulation) absolute width of the buffer may be more important than width relative to site-potential tree height. Thus comparing westside and eastside systems directly should be done with caution for these latter functions. Furthermore, the bars shown in Figures 14-2 and 14-3 should not be construed as representing the percent of function maintained. For example, most LWD is recruited within 30 m of the stream channel; consequently, in a westside system an unharvested buffer measuring one-half site potential tree may provide substantially more than 50% function with respect to wood inputs.

Stream Shading. Leaf Litter Inputs, Nutrient Regulation. Based on the review in the previous section, we conclude that buffer widths of approximately 0.75 site-potential tree heights are needed to provide full protection of stream shading, litter inputs, and nutrient regulation. FEMAT, PACFISH, and INFISH require riparian buffers along both fish-bearing and nonfish-bearing streams that are sufficient to protect these functions with the exception of intermittent and nonkey (PACFISH) and nonpriority (INFISH) watersheds in eastside systems. Eastside streams in nonforested areas may also be an exception because PACFISH and INFISH define buffer widths based on the 100-year floodplain; thus, the level of protection depends on whether the reach is constrained or unconstrained,

State forest practice rules generally provide less complete protection of shading, litter inputs, and nutrient control than do Federal standards and guidelines. In addition to having narrower buffers, the State forest practice rules allow activity within the riparian zone that may diminish riparian functions. For westside systems in California, buffer widths are sufficient to provide full protection of these functions only for fish-bearing streams (Class I) with side slopes exceeding 50%; buffer widths for all other States and stream classes are inadequate for maintaining full protection (Figure 14-2). For eastside systems in California, buffers are generally wide enough to maintain full function along fish-bearing streams with slope > 30% and steep (side slopes > 50%) nonfish-bearing tributaries that drain into fish-bearing streams, but not for streams on lesser slopes in each class (Figure 14-3). In addition, California allows substantial reduction in overstory conifers (75% removal), which may alter the composition of leaf litter as well as nutrient dynamics. Buffer widths for both fish-bearing and nonfish-bearing streams in western Oregon and Washington do not assure full shading, organic litter, and nutrient control functions, both because buffers are insufficiently wide and because removal of trees is allowed within the riparian zone (Figure 14-2). For eastside systems, however, these fixed-width buffers provide greater relative protection since site-potential tree heights are smaller compared to those in westside systems, Larger fish-bearing streams in Oregon (Type FI) appear to be fully protected, whereas medium-sized fish-bearing streams (OR FII) are marginally protected and small fish-bearing streams are less so. Idaho's forest practice rules provide buffers for fish-bearing streams that approach the fully protective width; however, 25% of existing shade may be removed. Washington's riparian buffers fur eastside fish-bearing stream are generally the narrowest of the four States, although they may be extended to 300 feet where riparian vegetation reaches that far from the active channel (Table 14-1). As with Idaho, 25% (or more if expected temperature increase is < 2.8°C) of canopy can be removed. Nonfish-bearing streams in Washington and Idaho receive little protection.

Bank Stabilization. Retention of riparian vegetation within 0.5 site-potential tree heights of the active stream channel appears necessary to maintain streambank stability. Buffers required by FEMAT,

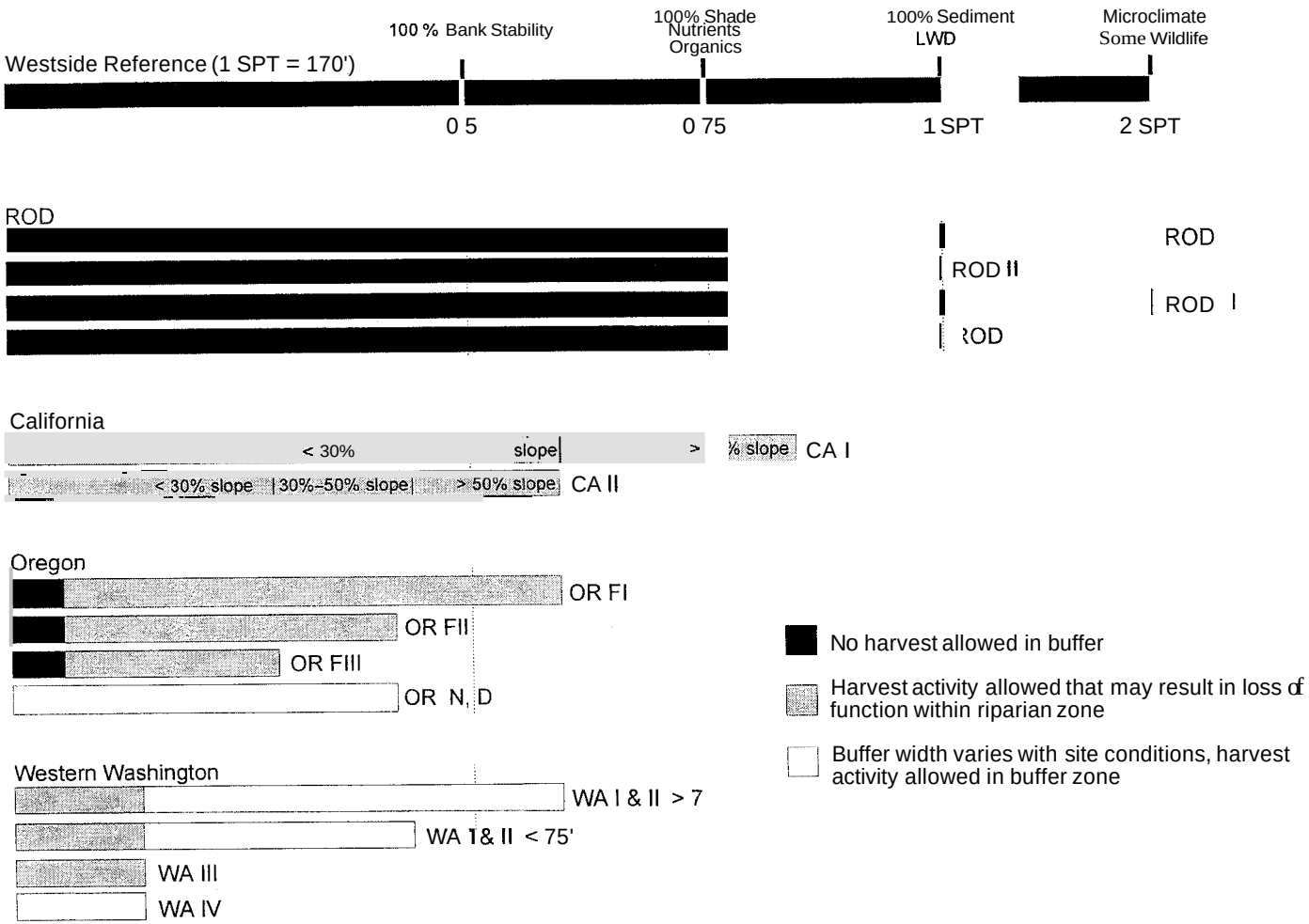


Figure 14-2. Riparian buffer widths by stream class for ROD and State forest practice rules for westside forests. Reference bar at top indicates approximate widths required for **full** protection of riparian functions.

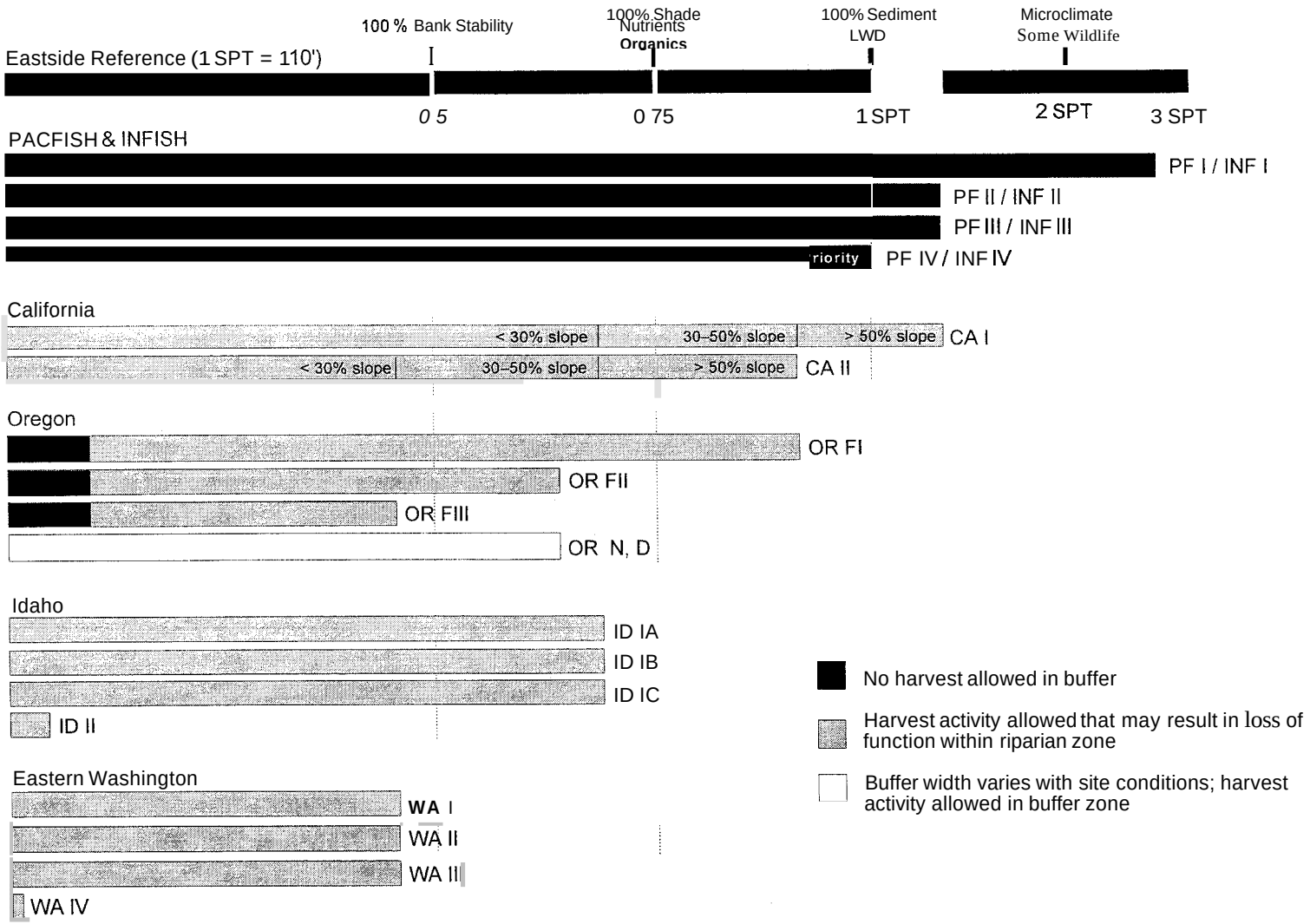


Figure 14-3. Riparian buffer widths by stream class for PACFISH and State forest practice rules for eastside forest. Reference bar at top indicates approximate widths required for full protection of riparian functions.

PACFISH, and INFISH for Federal lands generally provide full protection for this function along all fish-bearing and nonfish-bearing streams (Figures 14-2 and 14-3). Riparian buffers required by State forest practice rules are generally wide enough to protect bank stability along most fish-bearing waters in eastside systems. Little protection is provided for Idaho Class II and Washington Class IV waters. For westside systems, forest practice rules provide less coniplete protection of streanibanks, though we again note that absolute distance may be a more appropriate metric for evaluating effectiveness of riparian buffers in maintaining bank stability. Only Oregon's large fish-bearing streams and California's steep-sided (> 30% Class I, > 50% Class II), fish-bearing streams are well protected if buffers of 0.5 site potential trees are assumed to provide full protection. Because all States allow some harvest within the riparian zone, bank stability may be further compromised, although Oregon provides a 20-foot, no-harvest zone immediately adjacent to fish-bearing channels, which provides an additional measure of protection to bank integrity.

LWD Recruitment. Full recruitment of LWD by toppling, windthrow, or stream undercutting will generally occur if no-harvest riparian buffers of one site-potential tree height are retained. (Exceptions may occur in second growth stands where hardwoods have excluded regeneration of coniferous trees, leading to depletion of large size classes of debris). Riparian reserves provided by ROD, PACFISH, and INFISH are generally adequate to ensure close to 100% recruitment of LWD from riparian sources to both fish-bearing and permanent non-fish bearing streams on Federal lands, with the exception of intermittent streams in non-key watersheds of eastside systems (Figures 14-2 and 14-3). For nonforested streams on the eastside, the adequacy of riparian buffers for maintaining wood inputs varies depending upon valley and channel type, since riparian buffers are defined based on the 100-yr floodplain.

In contrast, buffers on private lands are generally inadequate to maintain full LWD recruitment to the stream channel, both because buffers are insufficient in width and because removal of conifers is allowed within the buffer zone (Figures 14-2 and 14-3). Only California Type I and II streams (side slopes > 30%) and Oregon Type FI streams require buffer widths approaching the dimensions needed for full recruitment and then only for eastside systems; however, long-term recruitment may be diminished by removal of conifers within the riparian zone. Murphy (1995) analyzed the effectiveness of State riparian buffers based on buffer widths and leave tree requirements along fish-bearing streams and concluded that the percent of LWD source trees remaining in the riparian zone after harvest in the four States are approximately 23% for California (Class I), 49% for Idaho (Class IB), 58% for western Oregon (Type FI), and 32% for western Washington (Type 1 & 2, < 75 feet width), if minimum standards are followed. These values indicate a substantial reduction in long-term ability of the riparian zone to provide wood to the stream channel under State forest practice rules. Botkin et al. (1994a) concluded that Oregon's Forest Practice Rules protect all riparian functions except that of supplying LWD, particularly large-sized pieces. Differences in each State's management allowances further influence the quality and quantity of conifer recruitment to streams. Oregon and California specify that leave trees must be conifers while Idaho and Washington permit hardwoods as well as conifers to qualify as leave trees. The lack of conifer retention will generally reduce the size and longevity of LWD that is recruited to the stream channel. Little protection is provided for recruitment of wood into nonfish-bearing channels. This wood is important in retaining sediments produced in headwater reaches (see below) and may be an important source of debris for downstream reaches if transported by high flows or debris torrents.

Sediment Control. Because mass wasting and channelized erosion are responsible for much sediment delivered to streams, management practices in upslope areas may be just as important as those used in the riparian zone. Considering only sediments generated by surface erosion within the riparian zone, buffers of approximately one site-potential tree would likely be effective in trapping most sediments, provided that slopes are not excessively steep (see above review). Under ROD, PACFISH, and INFISH, sediment retention is probably adequate for most streams except intermittent streams in non-key watersheds in eastside systems (Figure 14-2 and 14-3). State-required buffers are substantially narrower than those for Federal streams and as a consequence have a lower probability of providing full protection, although for gentle slopes buffers narrower than one site-potential tree may be sufficient to remove the majority of sediments. California is the only State that has explicit rules for increasing buffer widths based on slope steepness; forest practice rules for Idaho indicate buffers should be wider where slopes are steeper but provide no specific dimensions for steeper areas. Effectiveness of State riparian buffers for sediment control is also influenced by specific requirements for retaining groundcover or downed wood, both of which can reduce the impact of management activities on sediment retention capability. California requires retaining a minimum of 75% surface cover within the riparian zone and

treatment (mulching, seeding, rip-rapping , chemical stabilizers) of large bare patches created by forest practices. Oregon requires all vegetation within 10 feet and all trees within 20 feet of the stream channel be retained (except as allowed for road construction, yarding corridors, or stream crossings); in addition, Oregon requires operators to leave all downed wood within the riparian management zone along fish-bearing streams. Idaho forest practice rules state that logging and yarding within the stream protection zone of Class I streams should be conducted in a way to "minimize stream bank vegetation and channel disturbance" and to ensure "[sediment] filtering effects are not destroyed" but does not provide specific criteria for meeting these objectives. Similarly, Washington requires that logging and cable yarding within the riparian zone be conducted with "reasonable care" so as to minimize disturbing soils; use of tractor and wheeled skidding systems within the riparian zone must be approved by the Department of Natural Resources.

Based on site-potential tree heights, State forest practice rules would appear to provide somewhat greater protection for eastside streams than westside streams; however, this is probably not the case. The ability of riparian buffers to retain sediments is likely more a function of absolute distance (and slope) than distance relative to site-potential tree heights. Furthermore, overland flow likely occurs more frequently in eastside systems because forests are more open and the amount of organic duff and vegetative groundcover is typically less. State forest practice rules generally provide minimal protection for intermittent and nonfish-bearing streams. Yet these streams are extremely important in controlling sediment delivery because of their greater density (over 50% of the total length of stream channels in a watershed, Reid and Ziemer 1994, in Murphy 1995).

Other Riparian Functions. Riparian buffers required on Federal lands by ROD and PACFISH provide some protection of other riparian characteristics, including riparian microclimate, site productivity, and some riparian-dependent wildlife species, although degree of protection for these functions is uncertain. The level of protection is greatest for ROD Class I and III waters, PACFISH Type I streams, and INFISH Type I streams, which require buffers a minimum of two site-potential trees in width. No State regulations have fixed widths to address these additional functions. However, all States indicate that wildlife resources must be considered in planning timber harvest activities, particularly where sensitive species are potentially affected. California's forest practices rules specifically list microclimate modification as one potential wildlife concern to be evaluated. In addition, some States have snag (ID), downed wood (OR), or wildlife reserve tree (WA) retention requirements designed to protect certain wildlife needs. Oregon encourages retention of vegetation along small streams (including non-fish bearing) to protect amphibians that may inhabit these reaches, and Washington requires maintaining conifer/hardwood ratios similar to natural vegetation communities along fish-bearing streams, in part to protect wildlife values.

Summary and Conclusions

As noted above, specific recommendations for riparian buffer widths can only be made with a clear definition of riparian management goals. If the goal is to maintain *instream* processes over a relatively short time frame (years to decades), then fully protected riparian buffers of approximately one site potential tree (30–45 m in most Pacific Northwest forests) are likely adequate to maintain 90%–100% of most key functions, including shading, LWD recruitment (excluding wood recruited from upslope and upchannel areas), small organic litter inputs, nutrient regulation, and sediment control (for surface erosion in the riparian zone only). If the goal is to maintain natural microclimatic conditions within the riparian zone as well as large wood for nurse logs and nutrient contributions—conditions that may be essential for long-term (decades to centuries) maintenance of natural species composition and production of riparian vegetation as well as a number of wildlife species—then buffers need to be substantially wider. Similarly, prevention of blowdown within the riparian zone requires buffers of greater width. Cederholm (1994) has suggested that if the goal of management is to protect riparian ecosystems, there is a need to first define riparian areas from a functional perspective, and then maintain buffers around these ecosystems.

Based on the above review and analysis, we conclude that Federal riparian reserves outlined in ROD (FS and BLM 1994c), PACFISH (FS and BLM 1994a), and INFISH (FS 1995) in general provide adequate protection to riparian processes critical to maintaining salmonids in most instances. In addition, these riparian reserves provide some protection to microclimatic conditions within the riparian zone, help maintain recruitment of wood into the riparian zone, and provide greater protection for other riparian-dependent wildlife species along fish-bearing streams than do State forest practice rules. Protection for these latter functions is less along nonfish-bearing streams. In contrast, State forest practice rules do not ensure 100% protection for most critical riparian functions. Buffer widths are in most instances sufficient to protect bank stabilization and leaf litter inputs, but insufficient to provide 100% of LWD

recruitment, shading, and perhaps sediment control. In addition, the allowance of timber harvest within the riparian zone further diminishes the capacity of the riparian zone to provide all of these functions,

Because of the critical condition of many wild salmonid populations, we recommend that management activities be avoided within the riparian buffer zone under HCPs or other conservation agreements, particularly in old-growth and late-successional forests. Riparian forests that have not been disturbed by land-use activities provide the greatest level of protection for aquatic habitats and should generally not be disturbed until a significant percentage of riparian areas across the landscape has been restored. In second growth forests, particularly where natura1 vegetation has been replaced with hardwood trees and shrubs, management in the form of hardwood removal, thinning of small-diameter conifers in crowded condition, and planting of conifers may help accelerate the recovery of riparian forests, particularly with respect to recruitment of large wood (Berg 1995). These activities should be performed carefully so as not to diminish other riparian functions, including shading, sediment control, and bank stabilization. The overall goal should be to restore the riparian zone to a "natural" condition, not to maintain timber production within the riparian zone over the long term. For other land uses, including grazing and agriculture, riparian conditions likely bear little resemblance to natural conditions. In these areas, activities that are contributing to riparian degradation should be curtailed or avoided to allow these systems to recover. Where possible, efforts should be made to restore and reclaim wetland and floodplain areas that have been separated from riverine systems.

Although protection of riparian areas is essential to the conservation of salmonids, it is important to reiterate those functions for which riparian buffers have limited utility. These include hydrologic changes caused by alteration of upland vegetation and soil conditions in the catchment; sediment inputs from mass wasting and channelized erosion; nutrient or pollutant inputs that result from catchment modification or that reach the stream by channelized flow; and recruitment of large wood via processes other than toppling and windthrow. Consequently, riparian buffers should be viewed as one element of an overall watershed management plan. These buffers will only be effective if steps are taken to minimize cumulative impacts from upland areas as outlined elsewhere in this document.

14.2.4 Water Quality

Key Issues

High water quality is important not only for protecting salmonids and other aquatic organisms, but for preserving other beneficial uses as well, including recreational values, and agricultural, industrial, and domestic water supplies. Deterioration of water quality due to land use activities diminishes each of these values.

Water temperature influences all aspects of salmonid physiology, behavior, and ecology. Temperatures approaching or exceeding the physiologically tolerable range can cause direct mortality or acute stress in salmonids. In addition, relatively small increases in stream temperature at any time of year can adversely affect salmonids by changing metabolic requirements, behavior, rate of development of embryos and alevins, migration timing, competitive interactions, predator-prey interactions, disease-host relationships, and other important ecological functions (reviewed in Sections 4–6). Changes in both physiological and ecological processes may also occur with increases in diel temperature fluctuations. These adverse effects may occur even when temperatures are well within the physiologically tolerable range for the particular species. Because salmonids are adapted to the specific thermal regimes encountered throughout their life histories, maintaining natural temperature regimes is critical for their protection.

Salmonids require high levels of dissolved oxygen (DO) throughout most of their life stages with early life stages being most sensitive to reduced DO levels (reviewed in Section 5.2). Dissolved oxygen may be lowered in streams and rivers as a result of industrial and municipal discharges, nutrient-induced algal blooms, temperature increases, and increased siltation, which hinders exchange of water between surface and intragravel waters. Low DO levels influence developing eggs and alevins in a number of ways including reduced survival, retarded or abnormal development, delays in time to hatching and emergence, and reduced size of fry. In juveniles and adults, low DO impairs swimming performance, reduces growth, and inhibits migration.

Salmonids can also be adversely affected by a variety of toxic pollutants. These contaminants can enter streams as chronic inputs, such as industrial effluent or runoff from agricultural and mining areas, or as episodic inputs, such as chemical spills during transportation or failure of containment structures. Effects vary depending upon the chemicals, exposure, and interactions with other chemical, but can range from direct mortality and behavioral or morphological abnormalities to bioaccumulation of substances in tissues, making fish unfit for human consumption.

Recommendations

We recommend that HCPs and other conservation efforts include a strategy for maintaining levels of

temperature, DO, nutrients, and other dissolved materials within the natural range of variability for the particular body of water and time of year. Development of such a strategy will be most effective if it is preceded by a thorough assessment of current conditions within the watershed. This assessment should have three goals: to identify acute water quality problems within the watershed (e.g., areas where temperatures or DO levels violate State criteria or the tolerable range for extant salmonids during a particular life stage), to identify specific factors that contribute to these problems, and to compare current temperature regimes with reference conditions derived either from either historical data or data from relatively undisturbed watersheds within the region. In some watersheds, data for establishing appropriate reference standards for water quality parameters will be lacking. In these instances, reference standards may have to be inferred based on knowledge of presettlement conditions compared with current land and water uses. Current conditions should not be used except in undisturbed watersheds. The regional monitoring strategy outlined in Chapter 15 would, over time, assist in developing reference standards.

Maps identifying water-body types and uses can be compiled from State agencies with responsibility for water quality and fishery resources. Water quality data are available from Federal, State, and Tribal records, as well as from ambient monitoring by the applicant, and then related to land uses in the watershed. Specific water quality attributes that should be examined include temperature, dissolved oxygen, nutrients, turbidity, acidity, alkalinity, heavy metals, and other toxicants if there is reason to expect they may be entering aquatic ecosystems. Detailed analytic procedures for temperature, dissolved oxygen, turbidity, and nutrients are given in the *Federal Ecosystem Analysis Guide* (REO 1995). The Federal guidelines for temperature generally address only summer maximum temperatures. We recommend that analysis of diel temperature fluctuations and winter temperatures/ice formation also be conducted using historical records, comparisons of sites in perturbed versus unperturbed systems, and local knowledge. Where salmonid spawning occurs, monitoring of intragravel oxygen dissolved oxygen during the incubation period can help identify water quality problems, though sedimentation and bedload movement may also lead to low dissolved oxygen on salmonid redds.

Evaluation Criteria

The primary objective of the Clean Water Act (CWA) is to "restore and maintain the chemical, physical, and biological integrity of the Nations' waters." To this end, CWA directs States to establish water quality standards that describe beneficial uses of water in each drainage basin, numeric and narrative criteria necessary to protect these uses, and various policies to be implemented when managing State water quality (REO 1995). The Endangered Species Act (ESA) requires that any activities authorized by Federal agencies (including HCPs and other agreements) cannot jeopardize listed species.

We recommend that HCPs and other conservation efforts consider how new activities may adversely affect water quality in water bodies containing threatened or endangered species. In areas where existing water quality problems are impairing ecological function, conservation plans should seek to alleviate the causes of water quality degradation and maintain all water quality parameters within the range required for specific species and life stages. Conservation measures will be most effective if they are designed not only to ensure compliance with State water quality criteria but to maintain or restore water quality parameters to natural background levels.

Temperature. We believe that it is important *to* consider three fundamental questions in evaluating potential effects of temperature alterations on salmonids: Do temperatures exceed the maximum tolerable level for the particular species? Are temperatures within the preferred temperature range during each specific life stage? And do temperatures depart significantly from the natural range of variability for the particular body of water? This latter question is critically important because of local adaptation of individual salmonid stocks to the specific thermal regimes in their spawning and rearing streams. The importance of local adaptation to thermal regimes was highlighted by The Technical Advisory Committee of the Oregon Department of Environmental Quality (ODEQ 1995), who concluded that "It is not desirable to homogenize the temperature regimes of Oregon rivers if we want to preserve [life-history] diversity."

State water quality criteria generally consist of two components: an absolute numeric criterion for maximum summer temperatures (usually defined as the average daily maximum temperature over some defined time period) and maximum allowable increases (or decreases) for individual point sources or nonpoint source activities. Some States have maximum thresholds that vary depending on the presence or absence of particular species, with lower criterion in waters used by salmonids for spawning and rearing. In addition, maximum criteria in some States (OR, CA) vary with drainage basin or region in order *to* account for natural differences in temperature regimes. For example, under the proposed Oregon criteria, the lower Willamette and Columbia Rivers would be 20°C, 2.2°C higher than for the rest of the State (see below). Similarly, some

States have varying criteria for allowable increases, depending on a classification of the water body. For example, Washington water quality standards allow greater temperature increases in high-elevation waters. State water quality criteria, therefore, primarily target the first two questions listed above. Although minimizing the incremental change associated with a given activity somewhat addresses the need to maintain natural temperature regimes, it fails to prevent cumulative effects of multiple activities that may raise temperatures several degrees above natural levels, to the detriment of salmonids.

The available evidence suggests that most salmonids stocks are adversely affected by temperatures above 15.6–17.8°C; although fish may survive these warm temperatures, populations typically do not thrive under such conditions. The ODEQ (1995) recommended an absolute maximum criterion of 15.6°C for all waters, measured as the 7 day average daily maximum; a maximum threshold of 12.8°C for waters used by salmonid species for spawning and rearing; and a maximum threshold of 10°C for waters serving as habitat for bull trout. Based on an extensive literature review, Rhodes et al. (1994) recommended that no new activities that would increase water temperatures should be allowed on Federal lands where summer maximum temperatures exceed 15.6°C in waters that presently or historically supported spawning and rearing salmonids listed as threatened or endangered. We suggest that in evaluating HCPs for listed species or stocks, waters with maximum summer temperatures above 12.8–15.6°C should be considered potentially degraded, and that assessment of potential causes of degradation should be performed. Streams in certain regions (e.g., low-elevation, nonforested areas) may naturally experience temperatures exceeding these levels and thus are not necessarily impaired; however, temperatures above this range warrant a close look at potential human impacts.

Temperature tolerances of various salmonid species during each life stage have been fairly well-documented in the literature (reviewed in Tables 5-3 and 5-6). Figures 14-4, 14-5, and 14-6 summarize the temperature requirements of spring chinook salmon, coho salmon, and bull trout. We reconmend that these published ranges be used as a coarse screen for identifying temperature-related problems. If temperatures are above or below the preferred range, further assessment of potential anthropogenic causes is warranted.

Evaluating temperatures in relation to natural temperature regimes for the water body is more problematic. Ideally, reference standards should be established for each basin and water quality should be evaluated relative to those standards. Departures from reference conditions, even if State standards are not exceeded, would indicate potential impairment of aquatic ecosystem function. For example, if maximum stream temperatures exceed by more than 1–3°C those in **a** stream of similar size, elevation, and aspect in an unmanaged system, it may indicate the potential for indirect effects on salmonids. The difficulty lies in establishing appropriate reference standards, since few watersheds remain in undisturbed condition. Even streams in wilderness areas are subjected to grazing and may not be reliable indicators of natural temperature regimes. Therefore selection of reference sites and establishment of temperature standards should be a rigorous process. Sampled reaches must be randomly selected to ensure their representativeness and knowledge of all land uses upstream is needed. The ODEQ (1995) concluded that there is insufficient information to establish specific temperature requirements for the different stocks of salmonids and other cold-water fish in Oregon. The monitoring program outlined in Chapter 15 would aid in developing such standards.

In addition *to* the above temperature standards, we recommend that for all waters containing threatened or endangered stocks, no new activities be initiated that would result in measurable increases in stream temperature. This recommendation is consistent with the threatened and endangered provisions of the Oregon's proposed water quality standards recommended by the Technical Advisory Committee (ODEQ 1995). In addition, because of local adaptation of salmonids and the value of high-quality cold streams as habitats, we recommend against temperature criteria that allow greater anthropogenic warming in colder, high-elevation waters. Finally, we support the ODEQ (1995) recommendation that special protection be provided for coldwater refugia.

Dissolved Oxygen. Next to temperature, dissolved oxygen (DO) is the most frequently limiting water quality variable for aquatic life. State and Federal water quality criteria for salmonids vary with designated use, life stage, measurement, and statistic. For example, the criterion for intergravel DO needed in egg incubation varies from a minimum of 5 $mg \cdot L^{-1}$ and a 7-day mean of 6 $mg \cdot L^{-1}$ in Idaho to a minimum of 6 $mg \cdot L^{-1}$ and a 7-day mean of 11 $mg \cdot L^{-1}$ in the water column for Oregon. The EPA criterion is a minimum of 8 $mg \cdot L^{-1}$ and a 7-day mean of 9.5, both measured in the water column. Washington does not distinguish between incubation and other uses. Idaho requires 6 $mg \cdot L^{-1}$ or 90% saturation for all other uses. Oregon mandates a 30-day mean of 8 $mg \cdot L^{-1}$, a 7-day minimum of 6.5 $mg \cdot L^{-1}$, and a minimum of 6 $mg \cdot L^{-1}$ for cold water communities. In Washington, waters are classified by their minimum DO as fair (4 $mg \cdot L^{-1}$), good (6.5 $mg \cdot L^{-1}$), excellent (8 $mg \cdot L^{-1}$), or extraordinary (9.5 $mg \cdot L^{-1}$).

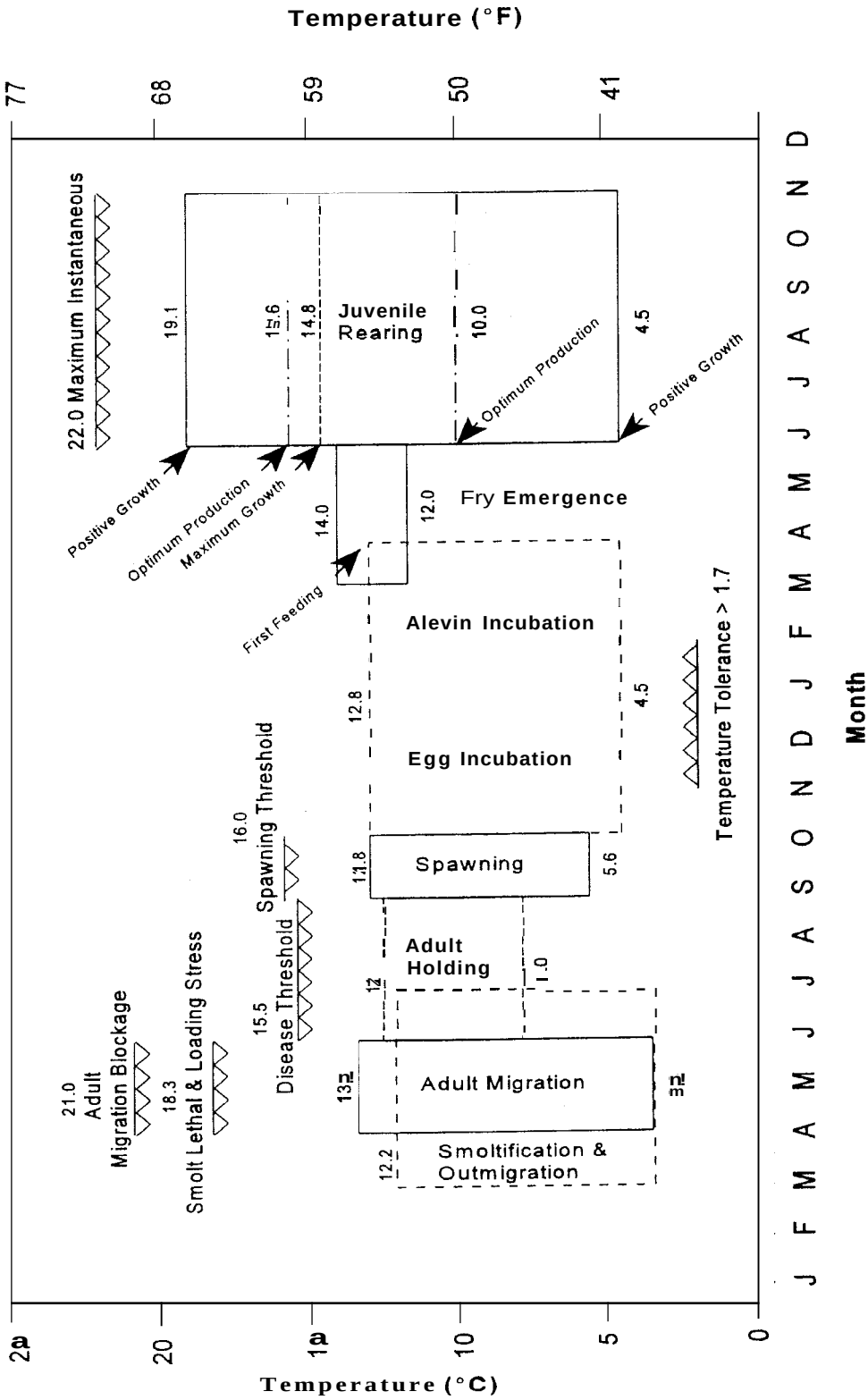


Figure 14-4. Spring chinook salmon temperature requirements by life stage. From D. McCullough (ODEQ 1995).

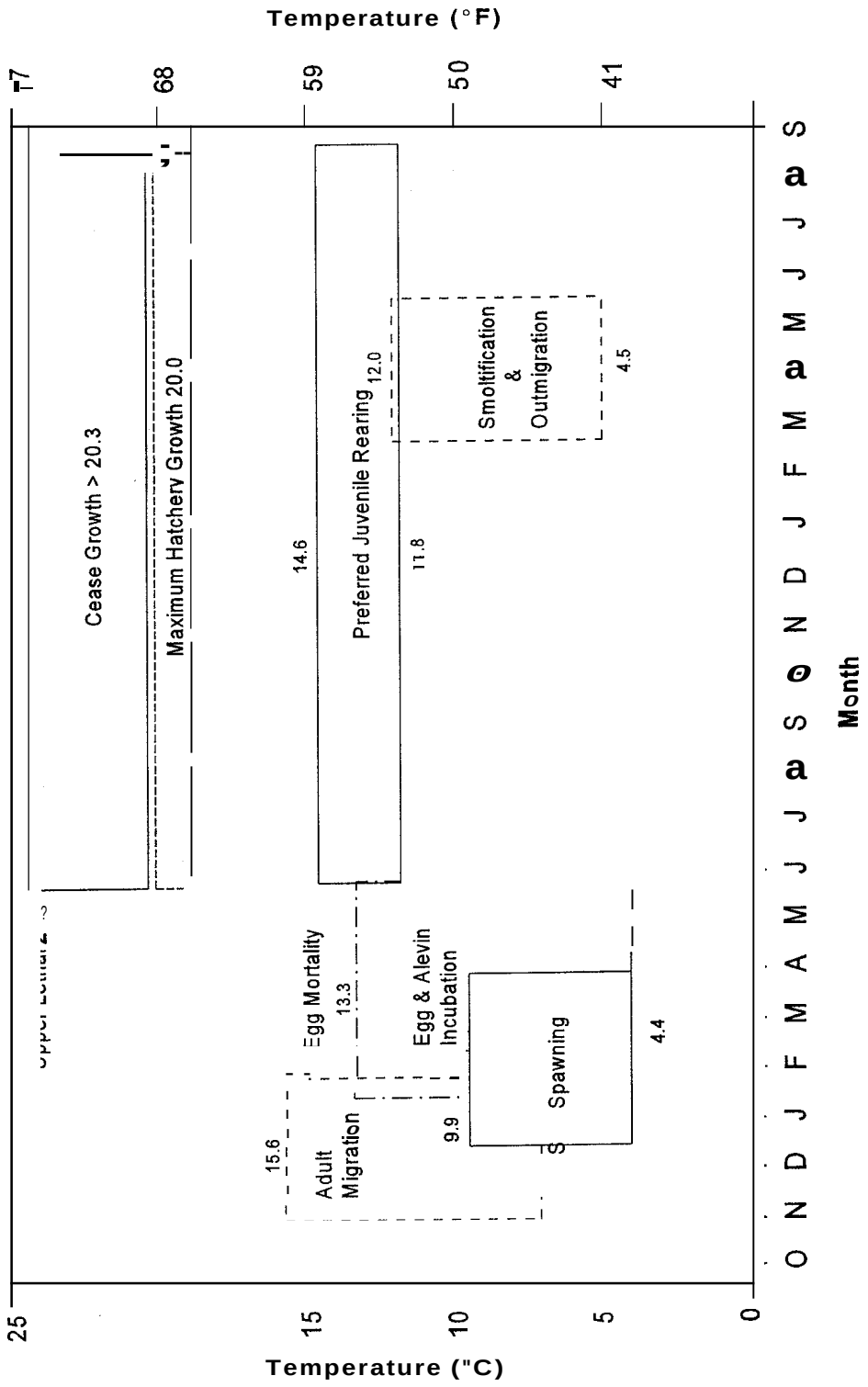


Figure 14-5. Coho salmon temperature requirements by life stage. Modified from ODEQ (1995).

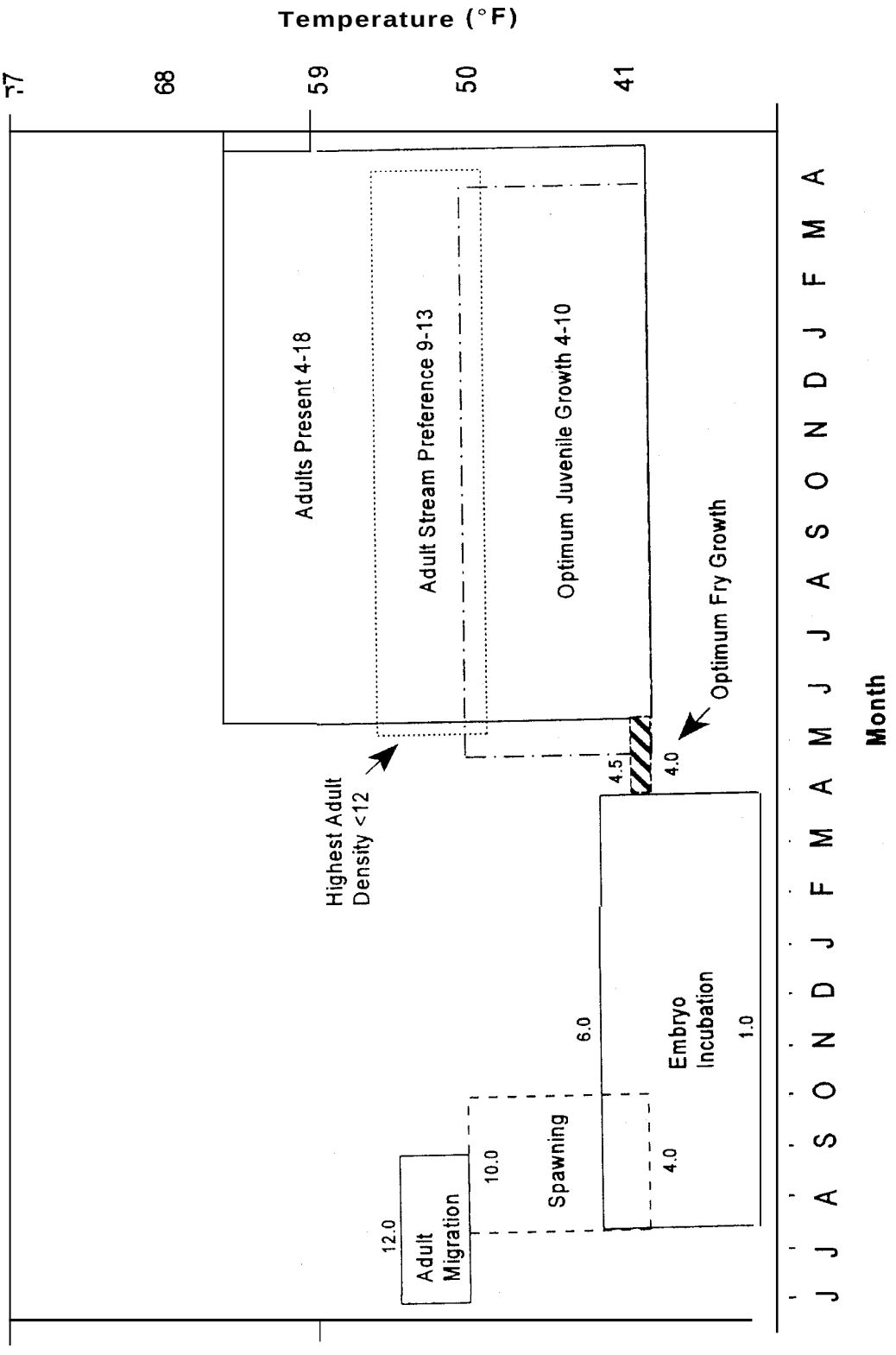


Figure 14-6. Bull trout temperature requirements by life stage. From D. Buchanan (ODFW) in ODEQ (1995).

Laboratory and field studies indicate that intergravel DO concentrations < 8 $mg \cdot L^{-1}$ reduce survival and size at emergence of fry, and that embryo survival is negligible below 6 $mg \cdot L^{-1}$. Salmonid rearing, the next most sensitive life stage, is affected at DO levels < 8 $mg \cdot L^{-1}$, which decrease swimming speed and growth and alter insect emergence timing (affecting a critical food source). DO concentrations $\leq$6 $mg \cdot L^{-1}$ result in avoidance, reduced metabolic efficiency, mortality of sensitive invertebrates, and decreased salmonid harvest rates.

As with temperature, any change from natural DO concentrations places salmonids at greater risk. In addition, most of the studies upon which these results are based were derived from short-term laboratory research where cumulative effects from many other stressors were purposely eliminated. For both reasons, plus the fact that threatened and endangered species require extra protection, we recommend an intergravel DO concentration of 8 $mg \cdot L^{-1}$ measured as a spatial median minimum in egg pockets during incubation, For salmonid rearing, we recommend a 30-day mean of 8 $mg \cdot L^{-1}$, and a 7-day mean of 6.5 $mg \cdot L^{-1}$, both measured by continuous monitors with a 30 min. recording interval.

Nutrients. The principal problem with nutrients (nitrogen and phosphorus) in most salmonid waters is their role in promoting excessive algal growths in streams and lakes. The result in both cases is reduced DO when the plants respire in the dark, or when they die and decompose. Nutrient enrichment may result from municipal and industrial point sources, livestock wastes, and agricultural wastes. Excessive loading of dissolved or fine particulate carbon can also deplete DO. In all these cases, oxygen and temperature criteria should suffice in place of separate nutrient criteria. Two possible exceptions to this are the protection of estuaries and lakes from eutrophication and avoidance of ammonia toxicity. Although nutrient enrichment may be of minimal concern in streams, when the nutrients eventually reach ponded rivers, lakes, and estuaries these systems may become overloaded and depleted of sufficient oxygen for salmonids or other uses. This is another reason for basin-scale planning and waste load allocation, but here again low DO concentrations are among the early indicators of concern. Ammonia toxicity is another matter. Any measurable concentrations of ammonia are indicators of potential chronic or acute toxicity. Because ammonia toxicity varies considerably with temperature, DO, and pH (primarily), we cannot recommend a single value; instead we advise referring to the EPA criteria document (EPA 1986). We do recommend that applicants with agricultural, municipal, and industrial discharges provide data on ammonia.

Toxicants. There are hundreds of toxic organic chemicals, and even more with unknown toxicities, as well as many metals and metalloids that are toxic. However, with a few exceptions, these are unlikely to occur in most salmonid waters. Metals are likely to be a problem only in the vicinity of mines and municipal and industrial point and nonpoint discharges. Toxic organics are likely to occur in the same discharges, as well as runoff from agricultural lands. Where toxic substances are believed to be a problem, we recommend that HCPs or other conservation efforts include sampling of fish for analysis of toxic effect (See Chapter 15 and Table 15-2). Chemical concentration data from composite whole fish samples are appropriate. Simple ICP scans should suffice for metals; GC scans for particular organic toxics should be based on usage and discharge rates in a particular crop or industry instead of an entire scan. Because many pesticides now in use are short lived, the best indicators of potential problems are use rates and direct bioassessments (see Chapter 15).

14.2.5 Roads

Key Issues

Roads can contribute to aquatic habitat degradation in several ways. Roads are frequently the dominant human-caused source of sediments delivered to streams due to mass failures of cut and fill slopes and channelized surface erosion. In addition, both paved and unpaved roads result in more rapid routing of water to the stream channel, potentially increasing the magnitude and frequency of peak flow events, which in turn can result in downstream transport of LWD, scouring of the stream bed and banks, and other structural modification of the stream channel. Placement of roads near streams frequently necessitates construction of revetments, which simplify channels, alter hydraulic processes, and prevent natural channel adjustments. Finally, runoff from roads in urban areas can contain significant concentrations of substances that are toxic to fish.

Recommendations

For HCPs or other conservation efforts that encompass whole watersheds or significant portions of watersheds, we recommend that a watershed-level strategy for minimizing impact of roads on aquatic ecosystems be developed. (Such a plan would likely be excessive for small landowners; however, the Agencies may wish to consider road density in evaluating conservation plans.) The strategy should include a long-term transportation plan for the watershed, a maintenance schedule for all existing roads, replacement of road culverts that are inadequate to allow adult and juvenile fish passage

during both high- and low-flow events, and removal and rehabilitation of roads that are no longer needed or that are contributing to the degradation of sensitive salmonid habitats. Issues germane to road design, construction, and maintenance at the site level are discussed in Section 14.3.1.

Preparation of a strategy for minimizing impacts of roads will require information on the current distribution and use of roads within the watershed, identification of existing drainage and erosion problems, and identification of all stream crossings and culverts. Road distribution information can be obtained through aerial photographs, whereas identification of erosion problems or inadequate culverts will require field surveys. Maps showing all roads and any associated resource problems within the area covered by the conservation plan should be generated.

Evaluation Criteria

Evaluation of the effects of roads on aquatic systems is confounded by the fact that roading virtually always accompanies other land uses, making it difficult to distinguish between causal agents. Nevertheless, some studies linking aquatic habitat conditions to cumulative effects of roading have been published. Cederholm et al. (1981) reported increased sediments in salmonid spawning gravels in watersheds where roads exceeded 3% of the total land area. Dose and Roper (1994) examined historical and current description of stream channels in nineteen watersheds in southern Oregon and found significant changes in channel morphology (widening and shallowing) in most streams where road density exceeded 0.84 $km \cdot km^2$. In contrast, changes in morphology were not significant in unroaded wilderness areas and drainages with lower road density. In both of these studies, logging was the predominate land use and was likely an important contributor to the observed degradation. Although these studies are insufficient for developing specific targets for road density or percent roaded area, they suggest that roads may serve as a general indicator of human disturbance and habitat quality.

14.2.6 Salmonid Distributions and Status

Key Issues

The ultimate goal of habitat conservation plans is to ensure the long-term persistence and health of salmonid populations through protection of their habitats. This entails protecting habitats required for all life stages, including adult migrations, spawning, incubation, winter and summer and rearing for juvenile and resident fish, and juvenile migrations. Effective conservation planning at the watershed level depends on knowledge of the distribution of salmonids within the watershed, the capacity of different portions of the watershed to sustain salmonids during various life stages, and the relative health of these populations. Identifying areas important to salmonid production is critical to the development of specific management strategies and prescriptions.

Recommendations

We recommend that HCPs and other conservation plans include a thorough assessment of salmonid distributions and status within the planning area. The goals of this assessment should be 1) to identify all habitats accessible (existing or potential) to salmonids, 2) to document the distribution and abundance of wild salmonids by species and life stage (including threatened and endangered stocks), 3) to identify areas of high productivity or importance for specific life stages (i.e., "hot spots"), 4) to determine trends in salmonid abundance within the watershed, and 5) to document past and present hatchery introductions to waters within the watershed. This information, together with information generated from the analysis of channel conditions and physical habitat (see Section 14.2.7), can then be used to develop specific management prescriptions that protect relatively undisturbed habitats, avoid sensitive or biologically important reaches, and restore degraded reaches.

Information on the present and historical distribution and abundance of salmonids within the watershed may be obtained from State and Tribal agencies, past stream surveys, historical records, and local residents. Because utilization of many streams by salmonids is poorly documented, field sampling may be needed to confirm recorded data. Field sampling may be especially important to document spawning habitats, particularly those in small and ephemeral streams. Information on the use of particular stream reaches by salmonids and their relative productivity is most likely to be obtained from local biologists, although such information may not be readily available. In these instances, field surveys may be needed. Historical records (e.g., biological surveys, migrant trapping data) may be useful in determining the cause of salmonid declines, such as the loss of specific life-history types within a population (see e.g., Lichatowich et al. 1995). In general, estimates of population size are unlikely to be available. The most likely source of data for population trends is counts from State-operated traps or surveys (e.g., juvenile migrants, escapement estimates, redd counts) or counts at fish passage facilities at major dams.

Specific products of the analysis should include maps of species presence and distribution within the watershed, maps of habitat use by species and life history stage, descriptions of the current status of

populations in the watershed, descriptions of trends in abundance (when possible), identification of habitats used by threatened and endangered species or stocks, and narrative summaries of stocking history.

Evaluation Criteria

Because the purpose of this analysis is primarily to gather information, no evaluation criteria are proposed. Assessments of habitat condition are discussed in the following section.

14.2.7 Channel Condifionand Physical Habitat

Key Issues

Channel conditions and physical habitats of salmonids have been substantially degraded by land-use practices throughout much of the Pacific Northwest. Stream channels have been altered directly through channelization, revetments, stream cleaning, and dam construction, and indirectly through changes in hydrology, sediment loading, and large woody debris recruitment (reviewed in Chapter 6). In many instances, cumulative effects of numerous land-use practices have resulted in streams that lack structural and hydraulic complexity, pool and off-channel habitats used for rearing and refugia, and high-quality spawning gravels. Artificial constraints on stream channels, changes in hydrology and sediment loading, and the loss of large woody debris together have destabilized stream channels, making them more susceptible to scouring during high flows, further altering substrate composition. These changes in turn influence spawning and rearing habitats of salmonids, as well as production of invertebrates that salmonids require for food.

Recommendations

Because the physical habitat degradation most often results from changes in other watershed processes, measures designed to minimize changes in watershed hydrology, sediment loading, and recruitment of large wood are likely to result in improved physical habitat for salmonids. However, where channel conditions have already been degraded it may be necessary to apply more conservative measures to facilitate recovery and prevent further damage. Therefore, we recommend that HCPs include a watershed-wide assessment of channel and habitat conditions. The purpose of this assessment should be several fold: to characterize channel forms and geomorphic processes directly affecting channels in the watershed; to identify reaches that are sensitive to large variation in runoff, sediment supply, and large woody debris; to identify reaches that have been subject to human-caused and natural disturbances (e.g., land use, flow diversions, stream cleaning, splash dams, channel incision, channelization, floods, and wildfires) and, where relevant, the land use practices associated with those disturbances; and to evaluate the effects these disturbances have had on sensitive reaches and how long it takes sensitive reaches to recover from disturbances (REO 1994).

Characterizing channel forms and geomorphic processes involves mapping of hillslope and valley features in the watershed, including floodplains, terraces, estuaries, alluvial fans, streamside slides, earthflows, and debris-flow termini, lakes, dams, and glacial moraines. The identification of sensitive reaches entails identifying and evaluating the condition of alluvial valleys or other reach types that are typically important to salmonid production. To evaluate past disturbance events, data on streamflows, landslides, vegetation cover, and land use can be obtained. Agency records and interviews with local residents provide information about past human disturbances, including timber harvesting, splash damming, mining, grazing, water diversions, stream channelization, and other activities that have likely modified channel attributes. Field sampling should be conducted *to* characterize specific habitat attributes including channel width and depth, bank condition, substrate composition, LWD abundance and size, pool frequency and size, and presence of beaver ponds and off-channel rearing habitats. Procedures for performing channel assessments can be found in the channel condition and physical habitat modules in the *Federal Ecosystem Analysis Guide* (REO 1995) and the stream channel assessment module of the Washington watershed analysis manual (WFPB 1994).

In assessing habitat conditions, a number of habitat concerns related to specific life stages should be considered (Table 14-2). For adult migration, key objectives include identifying barriers to migration, assessing frequency and condition of holding pools, and identifying important cold-water refugia, particular for species such as spring chinook that oversummer within streams. For spawning and incubation, HCPs should address the availability and condition of spawning gravels (including intergravel dissolved oxygen), as well as evidence of erosion, scouring, and dewatering of spawning redds. Summer and winter rearing habitats for juveniles and resident fishes should be identified and their conditions documented; habitat concerns include diminished frequency, size, and depth of pools, loss of off-channel habitats, reduced structural and hydraulic complexity (e.g., LWD), elimination of beaver ponds, and loss of both summer and winter cover. Habitat issues related to juvenile migration include water quality and quantity, instream cover,

Table 14-2. Habitat concerns, by salmonid life stage, that should guide conservation efforts.

Life stage	Habitat concerns
Adult migration	Impassible or poorly designed culverts Impassible dams or diversions Impassible because of water quality (high temperatures, pollutants) Reduced frequency of holding pools Lack of cover in holding pools Reduced cold-water refugia
Spawning and incubation	Availability of spawning gravel Siltation of spawning gravels Erosion of spawning gravels Evidence of redd scour Evidence of redd dewatering
Juvenile rearing	Evidence of diminished pool frequency, area, or depth Reduced cover for summer rearing habitats Poor water quality (high temperatures, pollutants, low DO) Dewatering of stream reaches Reduced hydraulic heterogeneity Reduced invertebrate production Reduced pool frequency (winter refugia) Reduced off-channel rearing areas Loss of winter cover in substrate interstices (increased cobble embeddedness)
Juvenile migration	Poor water quality (high temperatures, gas supersaturation) Lack of instream cover Impassible barriers (physical, chemical, biological)

and migration barriers. Each of these habitat concerns should be discussed relative to historical and current conditions. The *Federal Ecosystem Analysis Guide* (REO 1995) includes a module for assessing physical stream habitats, with emphasis on needs of salmonids.

Evaluation Criteria

Evaluation of salmonid habitats is complicated by the fact that there is substantial natural variability in habitat attributes. This variability arises from differences in the rates of watershed processes (water, sediment, and wood delivery) as well as differences in channel morphological features that control the fate of those materials once in the channel, including stream gradient, channel width, degree of constrainment, and bed material. We believe that, for most habitat attributes, existing data are insufficient *to* justify numeric criteria for various habitat elements, partly because there are so few unmanaged systems remaining in the Pacific Northwest (especially nonforested systems) to provide appropriate reference points, and partly because methods of measuring and reporting habitat characteristics have rarely been consistent between studies. Nevertheless, published data on habitat attributes in unmanaged systems may provide coarse-level metrics for assessing whether specific stream segments may be in degraded condition.

Channel Type. Channel type is an important variable for stratifying data related to physical habitat (*e.g.*, pools, LWD frequency), channel conditions, channel sensitivity, and salmonid distribution information. Channel segments should be classified as erosional or depositional, constrained or unconstrained, and by stream gradient. No specific criteria are relevant since these attributes are determined entirely by landform.

Large Woody Debris. The frequency and volume of large woody debris within stream channels is influenced by a number of factors including stream size and gradient and the age and structure of riparian vegetation, which determine loading rates of large wood. Figure 14-7 illustrates the high degree of variability in the frequency of large wood pieces in relation to stream width for unmanaged systems in the Pacific Northwest. Bilby and Ward (1989) reported that the mean size (measured as diameter, length, and volume) of individual wood pieces increased with increasing stream width, but that the frequency of LWD pieces decreased with increasing stream size. They attributed these trends to the

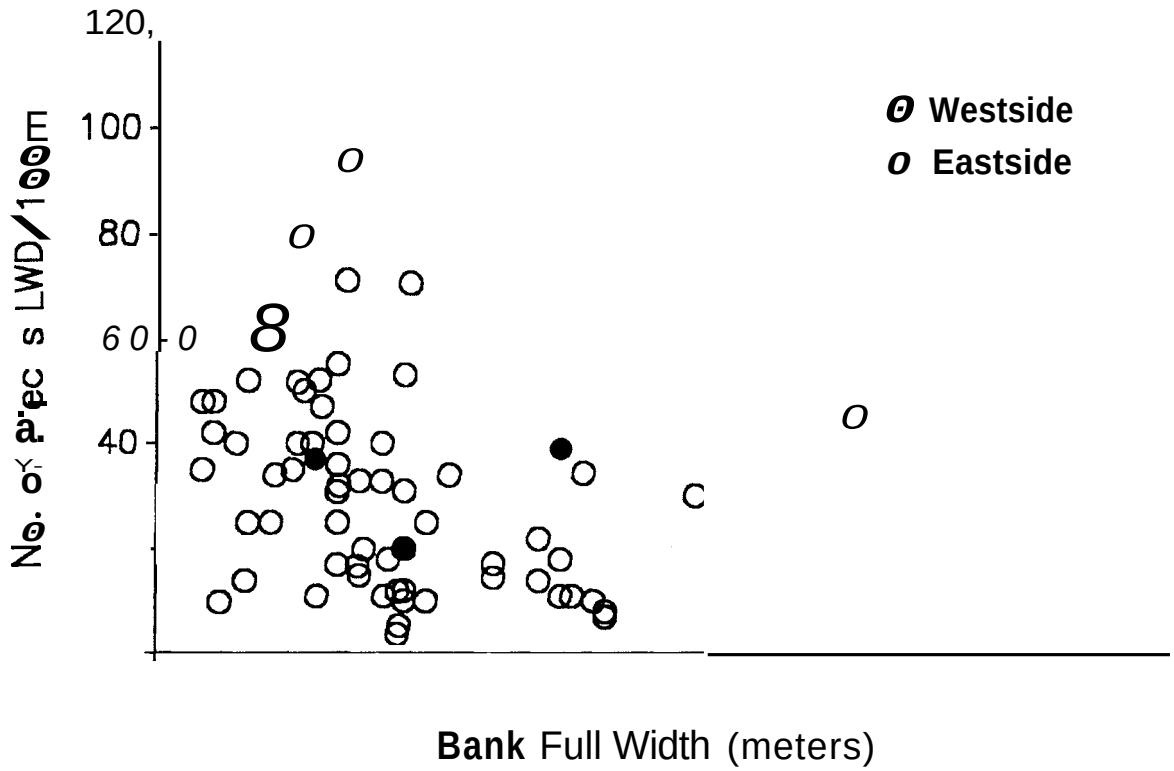


Figure 14-7. Abundance of large woody debris in relation to channel width for streams in the Pacific Northwest and Alaska. Data compiled from Robison and Beschta (1990), Cederholm et al. (1989), Murphy and Koski (1989), Fox (1992), Bilby and Ward (1991), Leinkamper and Swanson (1987), Long (1987), Fausch and Northcote (1991), Ralph et al (1991), Ralph et al. (unpublished data), and Dincola (1979).

greater stability of smaller size fractions in narrower streams, Other studies have indicated higher frequencies of wood in larger streams (e.g., Robison and Beschta 1990; Murphy and Koski 1989) or no trend in wood frequency with stream size (Ralph et al. 1991). Studies relating LWD frequency to stream gradient have been similarly variable. Murphy and Koski (1989) and Robison and Beschta (1990) found that LWD counts were highest in low-gradient (0.5%) reaches, but that at gradients of 1%-2.5% there was no consistent trend. Similarly, data from Sullivan et al. (1987) suggest no obvious trend in LWD frequency for gradients ranging from 3%-5%.

Peterson et al. (1992) stressed the need to establish target conditions for LWD in streams as a means of determining habitat condition. We concur that establishing targets is an important goal; however, in most instances data for developing such targets are generally not available. Peterson et al. **(1992)** recommended using regression equations developed by Bilby and Ward (1991) relating frequency and volume of LWD to stream width to set targets for LWD. These equations represent the most complete data available that we know of for Pacific Northwest streams; however, we suggest that they only be applied to the types of streams for which they were derived, namely forested streams in western Washington with widths of 4-19m and gradients from 1%-18%, and even then with considerable caution because of the high natural variation within this data set.

In addition, LWD counts alone may be poor indicators of habitat condition and effects of management. Ralph et al. (1991) found no difference in the number of LWD pieces between streams in managed and unmanaged forests, but that the average size of LWD pieces was significantly smaller in harvested systems. They therefore concluded that counts of LWD pieces alone are not useful as management objectives because they fail to account for important differences in the size (and therefore stability) of wood pieces. Bilby and Ward (1991) reported significantly higher frequencies of LWD in streams in old-growth forests than in second-growth or recently clearcut stands. **As** noted earlier, larger sized "key pieces" perform a critical function in retaining smaller debris. NMFS (1995) has developed provisional criteria for larger pieces of LWD for

streams in the Coast Range and east of the Cascades. They concluded that streams in the Coast Range should be considered "properly functioning" when they exceed 80 pieces/mile of wood larger than 61 cm (24 in) in diameter by 15.2 m (50 ft) in length, and where adequate sources for woody debris recruitment are present in the adjacent riparian zone. East-side streams are considered properly functioning where LWD exceeds 20 pieces/mile of wood greater than 30.5 cm (12 in) and 10.7 m (35 ft) in length and where adequate sources of recruitment exist.
The regression equations of Bilby and Ward (1989) indicating increasing average volume of individual debris pieces with increasing stream size reflects the greater ability of larger systems to transport smaller wood. This trend should not be construed to mean that larger pieces are unimportant in small streams or that large diameter trees could be removed without impairing ecological function. Pool area and sediment retention are both related to the size of wood pieces (Bilby and Ward 1989). In addition, small streams, even those without fish, may be important sources of LWD for downstream, fish-bearing reaches.

In developing LWD criteria, we therefore suggest that stream reaches need to be stratified by width, gradient, and ecoregion (or other indicators of vegetation type), and that both counts by size class and volumetric measures be employed. Rhodes et al. (1994) recommend against establishing specific numeric standards for in-channel LWD and instead recommend full protection of LWD recruitment from the riparian zone. We concur with the recommendation of Peterson et al. (1992) that a common definition of large woody debris be adopted throughout the region so that, over time, comparability of studies will be enhanced, allowing more meaningful targets to be defined. They recommended wood greater than 10 cm diameter by 2 m length be classified as LWD because the majority of studies have used this definition. We recommend that this definition be used to define minimum piece size for LWD but believe it is important to sytematically quantify larger-size fractions as well. The frequency distribution of different size classes is likely to be more important than total number (or total biomass) of all pieces exceeding some minimum length.

Pool Frequency and *Qualify.* Pool habitats are required by most salmonids at one or more life stages (reviewed in Chapter 5). The loss and reduction in quality of pool habitats has been identified as a major source of habitat degradation through large portions of the Pacific Northwest (McIntosh et al. 1994a). These alterations have resulted from removal and lack of recruitment of large woody debris, combined with increased sediment delivery to streams.

Pool formation depends on a wide variety of factors, including gradient, channel width, and LWD or other physical obstructions. Consequently, there is a high degree of natural variability in pool frequency and volume, even in unmanaged systems. Furthermore, methods for defining pool habitats have varied substantially among studies, making comparisons difficult. The most frequently used metric of pool habitats is the percent of total surface area in pools. Other indices include pool frequency, volume, and residual depth, the latter two of which may provide a better indication of pool quality than percent pool area or frequency. Pool frequencies for managed and unmanaged streams in Washington are shown in Figure 14-8 (see review in Peterson et al. 1992). In eastern Oregon and Washington, frequencies of deep pools ($>$ 1.6 m) in unmanaged systems ranged from 2.7 to 14.3 per kilometer of stream (B. McIntosh, USDA Forest Service, Pacific Northwest Station, personal communication).

Based on an extensive review of the literature, Peterson et al. (1992) recommended a target condition for percentage area of the stream surface area comprised of pools of 50% for Washington streams with gradients $<$ 3%. MacDonald et al. (1991) concluded that total area, depth, or frequency of pools may not always be a reliable indictor of anthropogenic effects. Because of the high degree of natural variation, we conclude that available data are inadequate to recommend specific criteria for pool frequencies that would be indicative of stream condition. The *50%* target recommended by Peterson et al. (1992) may be useful as a first indicator of potential degradation, but should not be widely applied outside of the region. NMFS (1995) has adopted provisional guidelines for pool frequency based on channel width. These are shown in Table 14-3. Again, we suggest that these values be used as general indicators, rather than absolute measures of habitat condition. The monitoring strategy suggested in Chapter 15 would produce consistent and reliable data from which regional targets could be derived.

Bank Stability. Erosion and slumping of streambanks can be an important cause of sedimentation and channel degradation in streams. Thus, bank stability can be a useful indicator of channel condition. However, we found no published information that would support establishment of specific numeric criteria for bank condition. Some bank instability is likely even in unmanaged systems. In wide alluvial valleys, lateral migration of the stream occurs through bank erosion and point bar accretion (MacDonald et al. 1991). In constrained reaches, temporary bank instability may follow the

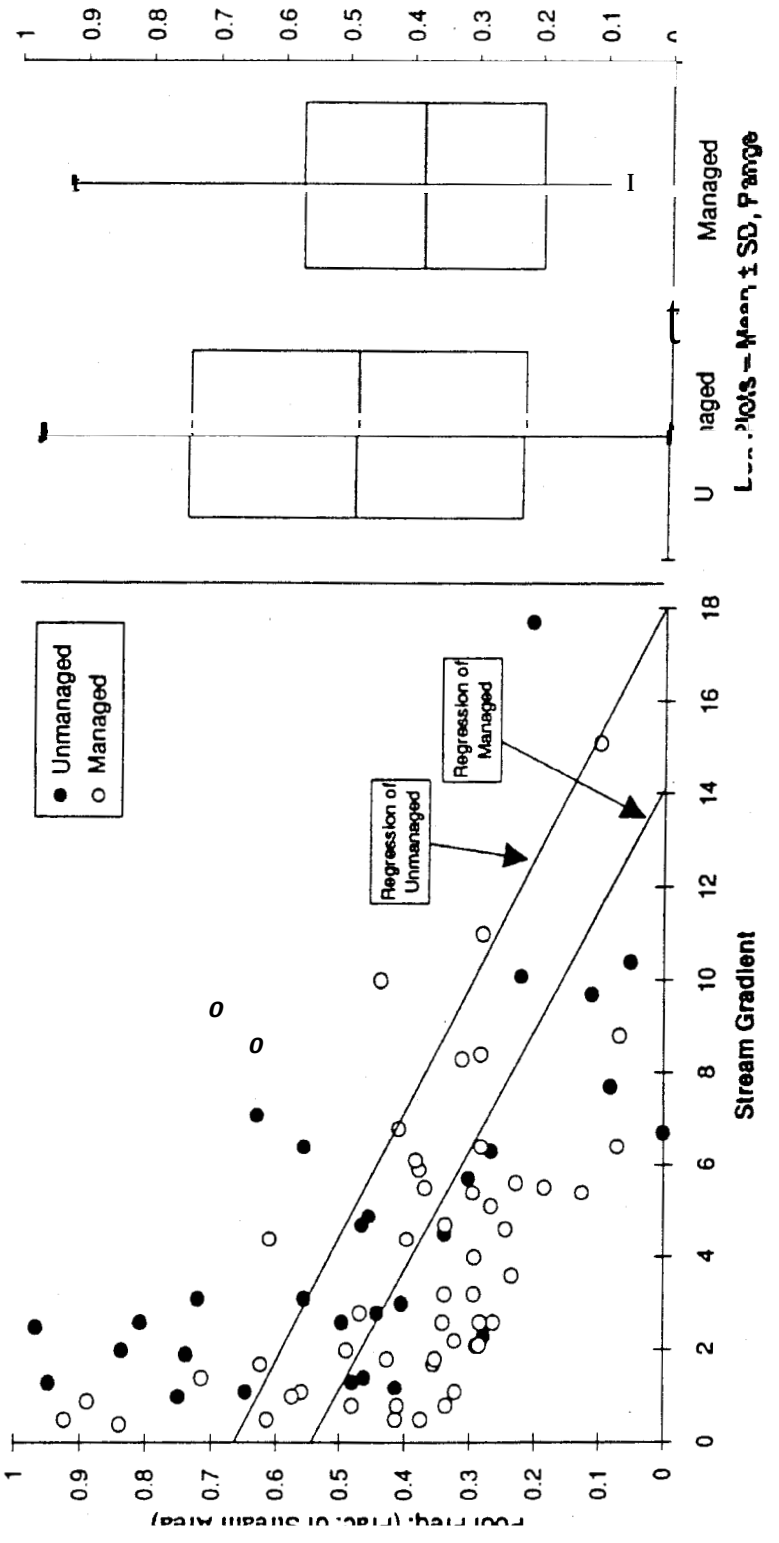


Figure 14-8. Relationship between fraction of the stream area comprised of pools and gradient for streams in managed and unmanaged forests in Washington. After factoring out the effect of gradient, pools area was significantly higher in unmanaged systems. From Ralph et al. (1991) in Peterson et al. (1992). Reprinted with permission of the author.

Table 14-3. Provisional minimum pool-frequency standards for determining properly functioning salmonid habitats. Proposed by NMFS (1995a).

Channel width (feet)	# Pools/mile
5	184
10	96
15	70
20	56
25	47
50	26
75	23
100	18

input of LWD that redirects hydraulic energy. Rhodes et al. (1994) and NMFS (1995) recommend that watersheds containing threatened and endangered species be managed so that 90% of streambanks are stable, although they provide no quantitative information to support this target. They suggest that for areas where this standard is not met, activities that would decrease stability of forestall recovery should not be permitted until the standard is reached or a trend of improvement is statistically demonstrated. We found no additional published information recommending criteria for bank stability, nor did MacDonald et al. (1991).

Substrate Composition. Excessive concentrations of fine sediments in spawning and rearing habitats can reduce survival of embryos and alevins by entombing embryos and reducing flow of dissolved oxygen, decrease the availability of interstitial hiding places, alter production of macroinvertebrates, and reduce total pool volume. A nuniber of different methods have been proposed for quantifying substrate composition and assessing the degree of sedimentation on substrate composition. For spawning gravels, fine sediments are commonly expressed as the percentage of sediments by weight or volume smaller than a particular particle size, usually < 0.85 mm or < 6.4 mm, two standard dimensions of substrate sieves. The effect of fine sediments of a particular fraction on incubating embryos and alevins depends on percentages of other size fractions (reviewed in Peterson et al. 1992), consequently, there can be substantial difference in results between studies. Bjornn and Reiser (1991) reviewed data from four laboratory studies and found that percent emergence of swim-up fry begins to decrease when percent fine sediment smaller than 2–6.4 mm (definition differed among studies) exceeded 15%. They also presented data for five salmonids indicating that embryo survival begins to decrease when percentage fines exceed 10%–25% (particle size < 6.4 mm), with rainbow and cutthroat trout being more sensitive than steelhead trout, kokanee, and chinook salmon (Figure 4.9 in Bjornn and Reiser 1991). Rhodes et al. (1994) concluded that survival to emergence for chinook salmon in the Snake River Basin is probably substantially reduced when fine sediment concentrations (< 6.4 mm in size) in spawning gravel exceed 20%. They recommended suspension of ongoing activities and prohibition of new activities where this standard is exceeded. Peterson et al. (1992) reviewed eleven laboratory and field studies of survival to emergence and concluded that in most instances an increase in percent fine sediment (< 0.85 mm in size) from 11% to 16% would result in a reduction in survival to emergence. Reported values were estimated by eye from figures and summary data from these studies.

Natural levels of fine sediment in spawning gravels vary with gradient and underlying geology. In western Washington, percent fine sediments (< 0.85 mm in size) in spawning gravels in unmanaged basins have been reported to range from 6.4% to 14.5% (reviewed in Peterson et al. 1992). Based on this review, Peterson et al. (1992) proposed a target of 11% fine sediments in spawning gravels for low-to-moderate gradient streams in Washington. They noted that this target should not be indiscriminately applied across geologic boundaries and that higher levels do not necessarily indicate degraded conditions. Rather, they suggest that where sediment levels exceed this target, the potential causes of sedimentation should be thoroughly examined. We concur with these recommendations.

Cobble embeddedness has frequently been used as an indicator of the quality of over-wintering habitat for juvenile salmonids, which hide in coarse substrate interstices during periods of low temperature and are adversely affect if these spaces become filled with sediment. MacDonald et al. (1991) provide a review of methods for measuring embeddedness in streams and difficulties associated with these methodologies. The State of Idaho is currently proposing embeddedness standards for protecting salmonid fry over-wintering habitat (MacDonald et al. 1991). These standards would call for cobble embeddedness not to exceed natural baseline levels at the 95% confidence level, where baseline levels are determined for unmanaged watershed with similar characteristics. Rhodes et al. (1994) recommended that watersheds should be managed so that cobble

embeddedness averages less than 30% in winter rearing habitats; however, they provided no empirical support for this particular threshold value. Peterson et al. (1992) suggested that an interstitial space index (ISI) developed by Vadenboncouer (1988) is more sensitive to change and bears a closer relationship to juvenile habitat requirements than cobble embeddedness; however, they concluded that data on interstitial space in Washington was lacking and therefore made no recommendations regarding appropriate standards. Because of the lack of available information, we make no specific recommendations regarding targets for interstitial space in rearing habitats. Nevertheless, monitoring of cobble embeddedness or interstitial space may allow detection of trends at a particular site (see Chapter 15).

14.2.8 Summary and Conclusions

Watershed-level planning has four important goals: 1) to address cumulative effects through time and space of multiple human activities and natural variation on aquatic habitats, 2) to assess current conditions within the watershed and identify existing resource problems, *3)* to relate existing resource problems to site conditions and land management practices, and 4) to use the knowledge gained to avoid future activities in areas that are sensitive to perturbations. Watershed analyses can also help identify and prioritize habitat restoration opportunities. In the preceding sections, we have reviewed specific processes that have been identified as important in affecting salmonids and their habitats. For ease of discussion, these processes were reviewed individually; however, it is important to recognize that upland, riparian, and aquatic processes interact in complex ways and that, consequently, conservation plans need to address all processes in a comprehensive and integrated manner. For example, improvements in large woody debris recruitment resulting from riparian buffers may be negated if peak flows or debris torrents increase in frequency in response to poor upland management. In addition, many factors may act synergistically to the detriment of salmonids. Lower stream flows, higher light levels (and photosynthesis), and warmer temperatures may combine to reduce oxygen levels in streams to levels that would not be reached by each factor alone. Similarly, the resistance of salmonids to disease is influenced by many water quality attributes (e.g., temperature, pollutants, oxygen levels). These examples represent only a few of the many possible ways in which multiple stressors may interact to produce effects greater than would be anticipated based on any single factor.

14.3 Site Level

The regional/basin and watershed-level analyses proposed in Sections 14.1 through 14.2 are designed to provide the context from which site-level prescriptions can be made that will effectively protect salmonids and, if desired, other resource values. Knowledge of existing watershed conditions and resource problems, as well as the potential sensitivity of different areas of the basin or watershed to land use activities, will enable owners of nonfederal lands to better avoid undesirable effects on aquatic ecosystems and the salmonids they support. Nevertheless, it is the cumulative effect of activities occurring at the site level that ultimately determine the health of aquatic and terrestrial ecosystems.

In this section, we briefly review specific management practices applied at the site level that afford the greatest protection to salmonids and their habitats. By site level, we mean the specific portions of the landscape upon which land-use activities are carried out by a landowner, such as harvest units, grazing units, agricultural fields, mining sites, and areas of urban development. We begin with a discussion of practices that are common to more than one land-use type and conclude by identifying practices specific to logging, grazing, agriculture, urbanization, and mining. We reiterate that the recommendations contained in the following sections assume that the affected watersheds support salmonids that are either listed or likely to become listed as threatened or endangered under ESA. Recommendations for protection of other species or resource values would likely differ. As specified in Section 10 of ESA, approval of an HCP requires that landowners discuss alternatives to a "taking" of a species that were considered and why these alternatives were not implemented.

14.3.1 General Practices

Riparian Buffers

Riparian buffers along all streams should be maintained, regardless of the type of land use. Specific dimensions of riparian buffers and management prescriptions will likely vary with site conditions and conservation objectives. **A** detailed discussion of riparian buffers can be found in Section 14.2.3. Aspects of riparian management relevant to specific land uses are discussed in subsequent sections.

Road Design, Construction, and Rehabilitation

Roads frequently constitute the dominant source of sediments delivered to streams. As discussed in Section 14.2.2, a long-term transportation plan for the watershed is desirable to minimize total disturbed area. Thus cumulative effects need to be considered when planning site-level 'activities.

It is beyond the scope of this document to recommend specific engineering standards for roads, however, we recommend the following general guidelines for road construction regardless of land

use type. Roads should be located away from streams, riparian areas, wetlands or other moist areas, and unstable hillslopes. Stream crossings should be avoided across or above reaches identified by watershed analysis as critical habitats for salmonid spawning. If crossings are unavoidable, they should be constructed in locations where the least amount of change in channel structure is needed and where potential for upslope impacts is minimal. Long-span skylines can be used to transport logs across steep-walled canyons, eliminating the need for creek crossings altogether, while minimizing construction costs. Culverts should be installed at angles and heights that allow passage during both high- and low-flow conditions. They should be placed below the original stream bed and have gradients less than 1%. Capacity should be sufficient to withstand 100-yr floods and care should be taken to ensure that water velocities in culverts are not excessive for fish passage.

New road construction should be minimized or avoided in areas where sediment-related degradation of salmonid habitats is identified in watershed analysis until the sources of that degradation have been alleviated. However, there may be instances where construction of new roads may reduce total sediment loads if it allows other, erosion prone roads to be retired and reclaimed. Construction methods for roads should seek to minimize the areal extent of soil disturbance. Landowners should adhere to minimum standards for width and gradient to reduce the amount of site disturbance. The height of cut slopes should be minimized to reduce the risk of failure, and materials should be end hauled rather than side cast where risks of slope failure are high. Areas disturbed during construction should be stabilized and reseeded following completion. Water needed for construction should not be withdrawn from streams bearing or upstream of habitats of threatened or endangered salmonids. Fuel should be stored away from streams and riparian areas, where the risk of contamination from spills is negligible. On slopes and soils where erosion potential is high, roads should be built only during the dry season.

Adequate drainage from road surfaces is critical to minimizing the erosive energy of water. Drainage control for new road construction should seek to 1) disperse, rather than concentrate, runoff: this can be accomplished using outsloped roads, cross drainage structures, and frequent relieving of drainage ditchlines; 2) avoid altering natural drainage patterns or discharging of water into non-drainage areas or fill slopes; 3) design drainage structures to withstand 100-yr-interval floods; 4) control scouring at culvert outlets using energy dissipators. All drainage ditches and culverts should be routinely maintained to prevent clogging with debris and sediments. Where drainage structures along existing roads are inadequate and causing erosion problems, these roads should be reconstructed with appropriate drainage or removed and reclaimed. **A** more complete list of recommendations for minimizing impacts of roads on aquatic systems is given in Table 8-1.

Active Restoration

Most of the recommendations in this document are designed to reduce or eliminate anthropogenic stressors that disrupt natural watershed processes and result in aquatic habitat degradation. These "passive restoration" techniques include such practices as 1) riparian buffers that preclude logging, grazing, agriculture and urban development; 2) cessation of irrigation withdrawals; 3) elimination of chemical use in farming, logging, and agriculture; and other practices that require no direct human intervention, other than alleviating the stress on the ecosystem. There are occasions, however, where direct mechanical, chemical, or biological intervention may be needed to accelerate the recovery of salmonid habitats or prevent further degradation. These "active restoration" techniques include such things as obliteration and revegetation of roads, removal or replacement of inadequate culverts or other barriers to migration, addition of logs or other structures to streams, removal of dams or rip-rap structures, gravel cleaning, vegetation manipulations (e.g., juniper removal, thinning, herbicide applications), use of prescribed fire, reintroduction of native species, and application of piscicides.

Kauffman et al. (1993) note that the greatest failure of many active restoration techniques occurs when these methods are implemented before the primary anthropogenic stressors have been eliminated. Furthermore, active restoration techniques frequently fail because factors limiting salmonid production are incorrectly identified. In each of these instances, costly restoration practices may fail to provide the presumed benefits to salmonids, or worse, may result in additional damage to stream ecosystems. Finally, many instream manipulations fail because the geomorphic context of a particular site is not considered.

Instream structural additions, in particular, have been widely employed throughout the west as a means of restoring structure to streams that have been degraded by past logging, splash damming, stream cleaning, mining, and grazing practices. Large sums of money have been devoted to instream restoration techniques, despite the frequent failure of structures to achieve desired biological outcomes or to withstand high flow events (Beschta et al. 1991; Frissell and Nawa 1992). Moreover, artificial structures can have significant negative effects on fish habitats. Hard structures can prevent natural channel adjustments, facilitate changes in channel morphology through changes in channel hydraulics (e.g., channel incision or widening), and exacerbate bank erosion and sediment inputs (Beschta et al.

1991). A common refrain in the literature related to active restoration, and instream manipulations in particular, is that these methods should be interim measures until natural functions can be restored; they should not be viewed as substitutes for or exemptions from habitat protection (Reeves et al. 1991; FEMAT 1993; Rhodes et al. 1994; Murphy 1995). We concur with this assessment. Placement of structures in streams should occur only as an emergency measure for preventing additional degradation, and then only after activities responsible for the degradation have ceased. Other active channel restoration techniques, such as reconnecting streams to off-channel areas, have greater potential for restoring salmonid abundance. These activities should be carefully planned and should not be considered substitutes for sound riparian management. Upland restoration techniques, such as erosion control programs, stabilization and revegetation of unused roads, and replacement of dysfunctional culverts have a higher likelihood of success with minimal risk to aquatic habitats.

74.3.2 Forest Practices

The impacts of forest practices can be reduced through a variety of practices (reviewed in Section 8.3). Emphasis should be given to minimizing the areal extent and intensity of disturbance to vegetation and soils. The site prescriptions discussed below provide high levels of protection for aquatic ecosystems.

Riparian Buffer Zones

Riparian buffers on all pernianent and ephemeral streams are recommended for protecting salmonid habitats. The specific dimensions of riparian buffers should depend on the specific ecological functions for which protection is desired (reviewed in Section 14.2.3). Once appropriate buffer widths are determined, we recommend that no forestry activities be allowed within these buffers in old-growth or late-successional forests. In second-growth forests, limited harvest, thinning, planting, or other manipulations may be appropriate in order to facilitate recovery and protection of key functions that have been identified through watershed analysis. These activities may be particularly appropriate in coastal forests where natural coniferous vegetation has been replaced by dense stands of alder and salmonberry, leaving little opportunity for conifer regeneration (Berg 1995). These activities should only be allowed when they can be performed without adversely impacting other riparian functions or values; use of ground-based equipment within the riparian zone should be avoided or minimized.

Silvicultural System

Rotation schedule in upland forests can be adjusted to minimize the total area in a disturbed state at any given time to minimize cumulative hydrologic effects (see Section 14.2.1). Sedimentation and soil compaction can be minimized if timber harvest, road construction, and site preparation activities are conducted during seasons of the year when potential for erosion is lowest. In most areas this will be the dry season; however, harvesting on snowpack may be effective in minimizing soil disturbance.

Harvest methods should be determined based on site-specific conditions. Logging should be avoided on areas identified in the watershed analysis as high risk for mass failures. In general, high risk areas will be those with steep slopes ($> 30°$) and unstable soil where there is a high probability that material will be delivered to the stream (see Section 14.2.2). Selective harvest, rather than clearcutting, is recommended for areas identified as moderately sensitive. Clearcutting is recommended only in areas of low sensitivity (i.e., low slopes, stable soils, far from streams).

Harvest System

Harvest systems should be determined based on site-specific conditions. On highly sensitive sites, helicopter logging minimizes disturbance to soils. Cable systems that partially or fully suspend logs off the ground (e.g., skyline) cause less disruption to soils than those where logs are not suspended (e.g., skidding). Use of ground-based equipment is advised only in low-risk areas.

For ground-based logging operations, designated skid trails can be established to minimize total area subject to compaction. Beschta et al. (1995) suggest that the percent compacted area can be reduced to 5% with careful planning. Careful planning of skid trails not only reduces soil disturbance but helps maintain high site productivity.

Site Preparation

Site preparation involves treatment of slash from logging operations and management of vegetation prior to planting. Appropriate treatment of slash depends on the specific resource concerns at the site. Where sediment delivery to streams, compaction of soils (by equipment used), and retention of nutrients on site are concerns, we recommend against burning of slash. Instead, we recommend scattering, mechanically chopping, or windrowing slash to control surface erosion. In some instances, such activities may be inappropriate if build-up of fuels would increase the risk of fires. Vegetation management entails removal of shrubs or trees by mechanical, chemical, and fire treatments. Mechanical treatments involving heavy equipment and scarification of soil should be avoided where sediment delivery and hydrologic alterations are of concern. Chemical treatments should be applied only outside of riparian buffer areas, including those of

headwater streams; for aerial spraying of herbicides and fertilizers, applications should be conducted to prevent drift into the riparian zone (apply parallel to riparian zone and under low wind conditions). Mixing of chemicals and washing of equipment should be conducted only where contamination of waters will not occur. Low-intensity prescribed fires may be appropriate in eastside forests for vegetation management.

Reforestation

To minimize the duration of hydrologic and erosion impacts, replanting of harvested areas should occur within two years of harvesting. Where reforestation occurs in the riparian zone, the goal should be to maintain natural vegetative assemblages in order to restore natural quantities, compositions, and seasonality of leaf litter inputs.

14.3.3 Grazing

Grazing impacts can be minimized by controlling livestock distribution, animal numbers, timing of forage use, kind and class of livestock, and total forage use, as well as by allowing complete rest from grazing (Platts 1991). The effectiveness of grazing strategies in protecting salmonids depends on the potential vegetation at the site; consequently, grazing strategies need to be tailored to the site and specific habitat concerns identified in the watershed analysis,

Riparian Buffer Zones

Riparian buffers are recommended for all permanent streams that support salmonids, as well as ephemeral streams that influence salmonid habitats downstream. The specific dimensions of riparian buffers should depend on the specific ecological functions for which protection is desired (reviewed in Section 14.2.3). We recommend that grazing be excluded in all riparian areas where function of riparian vegetation (shading, LWD, leaf litter inputs, sediment and nutrient control, bank stabilization) is currently impaired until such time as these functions are restored. This can best be accomplished by removing livestock or fencing of riparian areas. Once recovery has occurred, riparian grazing should be limited in duration and intensity to ensure these functions are maintained. Specific grazing strategies and their relative effectiveness in protecting aquatic habitats are shown Table 8-2. Only those with good-to-excellent ratings for all functions should be eniployed. Where riparian vegetation has been lost or reduced by livestock grazing, planting of native shrubs and trees is recommended to accelerate recovery.

Watering Facilities

Watering facilities should be located away from the stream channel and riparian zone, where possible. Where riparian areas are fenced, small access areas that allow livestock to take water directly from the stream may be appropriate where such access is not likely to degrade the stream.

Upland Grazing Strategies

Upland grazing should be managed to minimize surface erosion and disruption of hydrologic processes. Watershed analysis should identify portions of the range in poor, fair, good and excellent condition. Where range conditions are in other than good-to-excellent condition, we recommend temporary suspension of grazing until vegetation has recovered. Once conditions have improved, grazing strategies should be adjusted to ensure that conditions do not deteriorate again. This may be done by controlling grazing intensity by reducing the number or changing the class of livestock, reducing duration of grazing, or limiting total forage utilization (i.e., residual biomass).

Sediment Control

In areas where sediments are reaching the stream channel by surface erosion, steps should be taken to reduce surface erosion. Restoring vegetative cover (through control of grazing) should be given the highest priority. Where surface erosion is evident, mulching is recommended until vegetative cover is restored. Retentive structures may be appropriate for controlling rill and gullying erosion; however, design of these structures is critical, since poorly constructed dams or other devices may accelerate rather than alleviate erosion.

Chemical Applications

Application of chemical fertilizers and pesticides should be conducted to prevent contamination of waterways. No spraying should be conducted within the riparian zone or over surface waters. Aerial spraying should be conducted to prevent drift into the riparian zone (apply parallel to riparian zone and under low wind conditions). Mixing of chemicals and washing of equipment should be conducted only where contamination of waters is unlikely.

Channel Restoration

Where channel conditions have been degraded by grazing, replanting of riparian vegetation is recommended in order to accelerate recovery.

14.3.4 Agricultural Practices

Although specific methods for conserving salmonid habitats on agricultural lands are not as well developed, the principles for protecting streams on agricultural lands are similar to those for forest and grazing practices. Habitat conservation plans should emphasize protecting riparian zones, reducing sedimentation, minimizing fertilizer and pesticide inputs, and minimizing disruption of hydrologic processes.

Riparian Buffer Zones

Riparian buffers are reconmended for all permanent streams on agricultural lands that support salmonids, as well as ephemeral streams that influence salmonid habitats downstream. The dimensions of riparian buffers should depend on the specific ecological functions for which protection is desired (reviewed in Section 14.2.3). Use of agricultural machinery within the riparian zone or disturbance to vegetation or soils within the riparian zone should be avoided. Where channels have been degraded, by agricultural activities, planting of riparian vegetation native to the region is recommended. Conservation can be further enhanced by retiring converted wetlands from agriculture.

Sedimentation Control

Watershed analysis should be used to identify areas that are susceptible to surface erosion. Areas identified as highly erosive, with high probability of delivering sediments to streams, should be retired from agriculture. For moderately susceptible areas, various practices can be employed to reduce soil loss, including minimizing the area or frequency of tillage, mulching, use of cover crops during the rainy season, and terracing of hillslopes. Construction of settling basins in drainages susceptible to channelized erosion may further reduce sediment inputs.

Water Use

In circumstances where water has been over allocated or water quality issues identified, new water allocations should be approached with caution. This is particularly applicable where threatened or endangered stocks are present. All diversions of water from streams used by salmonids for spawning, rearing, or migration should be screened to prevent entrainment. For streams where water quality or quantity have been diminished by agricultural practices, a watershed conservation strategy should be developed to reduce the volume of water needed for agriculture, thereby increasing the amount available for aquatic resources. Components of this strategy should include one or more of the following: replacement of water-intensive crops with drought-resistant crops or crops appropriate for the precipitation regime within the region; elimination of water diversions; use of drip irrigation instead of high spray systems; lining of irrigation ditches; and maintenance of instream flows during critical stress periods (i.e., low flows, high temperatures). Where drainage ditches and tiles exist, intensive use of fertilizers or pesticides should be avoided because these structures are direct conduits to streams. In addition, drainage structures reduce summer water availability by routing water rapidly from the system and therefore should not be used unless combined with irrigation from deep groundwater.

Chemical Applications and Pest Control

Application of chemicals and pesticides should be conducted in a manner that minimizes contamination of aquatic systems. No chemicals should be applied within the riparian zone or over surface waters, and aerial applications should be conducted parallel to the riparian zone and under low-wind conditions to prevent drift into the riparian zone. Where water quality has been degraded by agricultural chemicals, organic farming and integrated pest management are recommended.

14.3.5 Mining Practices

Habitat protection measures for mining operations vary depending on the type of mining (e.g., surface mining, pit mining, underground mining, instream or floodplain aggregate mining). The goals of conservation practices at mining sites are similar to those of other activities (i.e., minimizing disturbance to soils and vegetation); however, the issue of potential contamination from toxic runoff and site reclamation also deserve special attention. The discussion below encompasses all types of mining, though not all HCPs will necessarily need to address each specific element.

Riparian Buffer Zones

We recommend that mineral or aggregate mining be avoided in streams or riparian areas of streams containing salmonids or that drain into salmonid habitats. Riparian buffers alone are likely inadequate to prevent chemical contamination of streams from untreated waste waters and runoff, thus, wastewaters should be treated before being released into streams (see below). Where channels have been degraded by past activities, active restoration including planting of riparian vegetation should be conducted.

Water Use

Water for mining purposes should not be withdrawn from streams supporting at-risk salmonids or habitats identified during watershed analysis as critical for salmonid production. Elsewhere, a water conservation strategy should be developed, including treatment and recycling of wastewaters and reductions in groundwater pumping where streamflow may be affected.

Sediment Control

Disturbance of soils is unavoidable during mining operations, however, care should be taken to minimize the aerial extent of ground disturbance. Lands that are denuded of vegetation should be stabilized as quickly as possible to reduce erosion, and methods such as contouring, mulching, and construction of settling ponds should be employed to minimize detachment and transport of soils. Disturbed sites should be revegetated as quickly as possible, and topsoil should be overlaid on mining

sites to assure successful regeneration. Where chemical constituents of mine spoils (e.g., pH, salinity, toxic metals) are likely to inhibit recovery of vegetation, spoils should be treated to ensure successful reestablishment of vegetation.

Water Quality

Mining should be avoided where tailings and wastewater have the possibility of entering aquatic systems. Wastewaters should be treated (acid neutralization, sulfide precipitation, reverse osmosis, electrochemical, or biological treatments) and recycled on site to minimize discharge to streams. Waters that are not clean enough to be re-used should not be discharged into streams. Pumping of groundwater should be avoided where lowering of the water table may facilitate transport of toxic materials. Control structures (barriers, ponds) should be constructed to retain toxic materials and should be built to withstand extreme precipitation events. Spoils containing toxic materials should be buried below the rooting zone of plants so that these materials are not taken up by plants and subsequently released into the environnient.

14.3.6 Urban Land Use

Urban land use poses the most difficult challenge to salmonid conservation planning, both because ownership is distributed among many individuals and because in most instances the landscape alteration approaches permanence. The most effective means for minimizing impacts is through county and city land-use planning.

Riparian Buffer Zones

Riparian buffers are perhaps even more critical in urban areas than in agricultural, range and forest lands because of the intensity of disturbance in surrounding uplands. Those riparian areas and wetlands that have not been paved or otherwise developed should be preserved and no new development allowed. Where feasible, impervious surfaces, such as parking lots and abandoned buildings, should be removed and vegetation restored.

Hydrology

Recommendations for minimizing the percent of landscape with impervious services is equally germane at the site and watershed levels (see Section 14.2.1). A program for reducing inipervious surfaces is currently being developed for Olympia, Washington, and should serve as a model for other urban environments (PWD 1995). Similarly, alternative forms of transportation (cycling, mass transit) should be promoted to reduce tlie need for additional roads. Where urban water withdrawals are degrading salmonid habitats, water conservation and recycling should be promoted. Further channelization of degraded streams should be avoided and wetlands should be maintained or restored.

Sediment Control

New construction of roads and buildings should be avoided on steep hillslopes that are susceptible to surface erosion and mass wasting. Sediment control measures, including matting, mulching, seeding, and construction of sediment traps should be employed at all new construction sites. Erosion can also be avoided by performing new construction during the dry season.

Water Quality

It is assumed that urban runoff is a major potential source of contaminants for salmonid-bearing streams, lakes, and estuaries. In such cases, urban stormwater should be routed through waste treatment facilities. In addition, use of chemical pesticides and fertilizers should be discouraged.

14.4 Data Needs

To perform the analyses outlined in the preceding sections, a substantial amount of information is needed, including data on ecoregion, climate, hydrology, geology, soils, stream channel networks, vegetation, disturbances (natural and anthropogenic), land use, and water use. Aerial photographs are particularly important in assessing historical and current watershed conditions. Potential data needs for watershed-level analyses related to physical and chemical processes are indicated in Table 14-4. In some instances, data are readily available in useable form from Federal or State agencies. Other data can be derived from existing data (e.g., slope stability will be based on topography, soil type, vegetation, etc.). Additional data are likely to be obtained only through field surveys and historical archives.

Data potentially needed for analyses of biological processes at the regional, basin, and watershed levels are listed in Table 14-5. Some of this information will already have been gathered for analyses of physical and chemical habitat attributes. Biological data needs include historical and current information on salmonid production; species distribution maps for salmonids, as well as other aquatic and terrestrial biota; distribution maps for threatened and endangered species of fishes and other taxa; species diversity maps; and genetic analyses. Some of this information can be obtained from Federal and State agencies, although in some regions, biological information may be sparse. Other data, including key watershed designations for private lands and ESU delineation for salmonids, are currently not widely available and it will be tlie responsibility of the agencies to develop this information for HCPs and other conservation efforts. A listing of sources for physical and biological data and how this information may be obtained can be found in Appendix A.

Table 14-4. Potential data needs for performing analyses of relationships between land-use practices and physical-chemical processes in watersheds, riparian zones, and streams. Data availability codes: 1 = maps or data usually available, 2 = poor coverage in some regions, 3 = maps or data generally derived from other data or field surveys.

		Hydrology			Sediment transport		Riparian functions					Channel condition			Water quality	
Data type	Data avail.	Peak flow	Base flow	Season. timing	Mass wast.	Surf eros.	LWD	Org. litter	Stream shade	Bank stabil.	Nutr. cycl.	Bed morph.	Substr. type	Phys. struct.	Strea m temp.	Pollu-tants
General maps																
Ecoregion maps	1	x	x	x	x	x									x	
WAU maps (WA)	2	x	x	x	x	x										
Topographic maps	1	x	x	x	x	x			x			x			x	
Aerial photographs																
Current	1	x	x	x	x	x	x	x	x	x	x	x	x	x	x	
Historical	1	x	x	x	x	x	x	x	x	x	x	x	x	x	x	
Climate																
Rainfall map	2	x	x	x	x	x										
Rainfall records	2	x	x	x	x	x										
Snowfall records	2	x	x	x	x	x										
Rain/snow zone maps	3	x	x	x	x	x										
Temperature records	2	x	x	x											x	
Hydrology																
Peak flow timing	1	x		x												
Streamflow records	2	x	x	x												x
Geology																
Bedrock map	1		x		x	x						x	x	x		
Structural geol. map	2		x		x	x						x		x		
Soils																
Soil type	2		x		x	x					x		x			
Soil descriptions	3				x	x							x			
Slope stability	3				x							x	x			
Stream channels																
Channel network	3	x	x	x	x	x										
Stream classification	2		x				x	x	x	x	x					
Vegetation																
Current veg. (type/age)	2,3	x	x	x	x	x	x	x	x	x	x			x	x	
Historic veg. (type/age)	2,3	x	x	x	x	x	x	x	x	x	x			x	x	
Potential vegetation	3	x	x	x	x	x	x	x	x	x	x			x	x	
Disturbances																
Wildfire	3	x	x	x	x	x	x	x	x	x	x		x	x		
Landslides	3				x		x					x	x	x		
Debris torrents	3				x		x					x	x	x		
Land uses																
Road maps (by age)	2,3	x	x	x	x	x						x	x	x	x	
Logging history maps	1	x	x	x	x	x	x	x	x	x	x	x	x	x	x	
Prescribed burns	?	x	x	x	x	x	x	x	x	x	x		x	x	x	
Grazing history	3	x	x	x		x	x	x	x	x	x	x	x	x	x	
Agricultural history	3	x	x	x		x	x	x	x	x	x	x	x	x	x	x
Mining history	3				x	x	x	x	x	x	x	x	x	x	x	x
Landfills	1					x							x			x
Urban area	1	x	x	x		x	x	x	x	x	x	x	x	x	x	x
Current land use	1	x	x	x	x	x	x	x	x	x	x	x	x	x	x	x
Water uses																
Dams	1	x	x	x								x	x	x	x	
Irrigation diversions	3		x	x								x	x	x	x	x
Point source discharges	3															x
Stream temp. records	2														x	

indicates agencies will most likely be responsible for data preparation

		Salmonid Habitat					Population Level					Community Level			Other Biota	
Data type	Data avail	Spawn	Incub	Sum rear	Wint rear	Migrat barriers	Pop viab	Life hist div	Meta pop	ESUs	Connec tivity	Comp	Pred	Disease a paras	Bio divers	Species at risk
General maps																
Key watershed maps	2						x		x		x					
Topographic maps	1	x		x	x	x										
Aerial photographs																
Current	1	x		x	x	x										
Historical	1	x		x	x	x										
Hydrology																
Streamflow records	2	x		x	x											
Stream channels																
Channel network	3										x					
Pool/riffle ratios	3	x	x	x	x							x	x			
Salmonid production																
Adult migrants	2						x		x							
Downstream migrants	2						x		x							
Population estimates (res)	3						x		x							
Historical records	3	x		x	x	x	x	x	x							
Species distribution																
Salmonids current	2								x	x	x	x	x			
Salmonids historic	2								x	x	x	x	x	x		
Native fishes	1											x	x			
Nonnative fishes	1											x	x			
Hatchery locations	1									x		x	x	x		
T&E fishes	1														x	x
Other aquatic T&E biota	1														x	x
T&E terrestrial biota	1														x	x
Biological surveys																
Fishes	2 3	x	x	x	x	x	x	x	x			x	x	x	x	x
Other biota	2,3											x	x	x	x	x
Species diversity																
ESU maps*	3														x	x
Genetic info (salmonids)	2						x		x	x						
Fishes*	3														x	
Other biota*	3														x	
Water quality																
Stream temperatures	2	x	x	x	x	x	x	x			x	x	x	x		
Dissolved oxygen	2,3	x	x	x	x	x	x							x		
Intergravel DO	3	x	x													
Turbidity	2,3	x	x	x	x	x										
Bacteria/pathogens†	3															
Acidity	2 3	x	x	x	x	x										
Alkalinity	2 3	x	x	x	x											
Toxic contaminants	2 3	x	x	x	x	x										
Land uses																
Road maps	2 3	x	x	x	x	x	x	x								x
Logging history maps	1	x	x	x	x	x	x	x								x
Grazing history maps	3	x	x	x	x	x	x	x								x
Agricultural history maps	3	x	x	x	x	x	x	x								x
Mining history	3	x	x	x	x	x	x	x								x
Urban area maps	1	x	x	x	x	x	x	x								x
Current land use	1	x	x	x	x	x	x	x								x
Water uses																
Dams	1	x	x	x	x	x	x	x			x	x	x	x		x
Irrigation	3	x	x	x	x	x	x	x			x			x		x
Domestic/agricultural wells	1			x												

† Bacteria/pathogen data for human health concerns and general indicator of water quality

15 Monitoring Salmonid Conservation Activities

Monitoring plays a critical role in all commercial, ecological, and social activities. It is the process that researchers use to obtain data and develop procedures through which a society assesses how well it is achieving its objectives. Ignorance of those objectives, or failure to adequately measure progress toward goals, guarantees they will not be met and increases the probability of undesirable consequences. This chapter presents monitoring elements that enable assessment of condition and detection of statistical trends in aquatic ecosystems at spatial scales from site to region. Sampling designs and indicators are proposed to track trends in physical, chemical, and biological conditions in uplands as well as riparian areas and streams so that critical planning elements can be monitored at appropriate spatial scales and temporal frequencies. Although there are many types of monitoring to obtain inforniation for many purposes, we focus on two major types: implementation monitoring and assessment monitoring (sensu FS et al. 1994). Planners and managers use implementation monitoring to determine compliance with the term of HCPs and other conservation agreements, and they, like scientists, use assessment monitoring to determine the effectiveness of activities in protecting or restoring salmonids and their habitats. Assessment and iniplementation monitoring are proposed both for individual HCPs and for providing the regional context to evaluate the overall effectiveness of salmonid conservation activities. Section 15.1 offers general guidelines for both types of monitoring programs. Specific issues for implementation and assessment monitoring are described in Sections 15.2.1 and 15.2.2, respectively. Sections 15.2.3 and 15.2.4 discuss the recommended sampling design and indicators.

15.1 General Guidelines for Monitoring Ecosystems & Salmonids for Conservation Planning

Because the Pacific Northwest now lacks an integrated approach for monitoring salmonids and aquatic ecosystems, we have difficulty determining whether changes in characteristics reflect fundamental changes in ecosystem function and structure, identifying the stressors associated with the changes, and quantifying the degree to which ecological problems are increasing regionally (Messer et al. 1991; Botkin et al. 1994). An effective program for monitoring salmonid conservation activities, as suggested in Chapter 10, would be long-term, multiscale, interdisciplinary, and interinstitutional. In addition to the above concerns, we offer four general guidelines based on our own experience and that of other monitoring programs.

15.1.1 Long-Term Monitoring

Monitoring over the long term documents trends in ecosystem conditions that occur in response to natural and anthropogenic disturbances, and it allows separation of the effects of human activity from natural variation. Over short time periods, natural variation in climatic conditions can produce strong signals that may mask anthropogenic effects. Furthermore, the effects of many human activities manifest themselves long after an activity has ceased, often in response to extreme environmental events (e.g., mass wasting associated with major storm events).

We recommend developing a common set of quantitative indicators for the Pacific Northwest and standardized methods of data collection. Annual monitoring (though not necessarily at the same sites each year) is best conducted by technically trained crews and ideally should continue for centuries. Issues important to successful implementation of a long-term monitoring program include ensuring adequate funding, scheduling of monitoring activities, archiving and retrieval of monitoring data, periodic reporting of monitoring results, and application of monitoring results to management situations (e.g., adaptive management). These issues are discussed in greater detail elsewhere in this chapter.

15.1.2 Multiscale Monitoring

Monitoring across many scales measures the effects of site- or reach-scale management activities as well as cumulative effects at the level of watersheds, basins, ecoregions, and multi-State regions. Monitoring crosses disciplines because ecosystems are complex aggregations of biotic and abiotic components, and those involved represent those areas of ecological expertise. Statistical

sampling designs used at both the population and site levels facilitate the conduct of monitoring at appropriate spatial and temporal scales. Conipliance can also be evaluated at local and regional scales to ensure that planned practices are implemented as outlined in conservation agreements across the region. When management practices are also monitored at local and Pacific Northwest scales, certain results can be determined: 1) the site-specific effects on salmon of conservation activities, 2) trends in regional distribution of salmon species and populations, and 3) the effects of salmon conservation on human society. A subset of indicators applied at the site, stream section, catchment, and region scales would facilitate data integration and analysis. If multiscale monitoring is allied with long-term monitoring over many decades or centuries, integrated observations about trends would amplify today's piecemeal knowledge about salmon populations, ecosystem conditions, land use, and the productivity of the lands for commercial resources.

Although this region presently lacks the program implementation and assessment monitoring suggested above, it does have many of the necessary pieces in place at the private, State, and Federal levels—at least for indicators. Differences in the perceived acceptability of qualitative versus quantitative indicators seem resolvable; however, fundamental differences in sampling designs hinder comparisons across institutions. As suggested by FS et al. (1994), a proposal to test instream and riparian indicators and designs could bring the Federal agencies and their cooperating State agencies closer together (Mulder et al. 1995).

15.1.3 Interinstitutional Monitoring

Monitoring becomes interinstitutional because lands are held by many different institutions, both public and private, and because many agencies have regulatory and management missions that directly or indirectly relate to salmonid conservation. Given the roughly 200,000 stream miles and 400,000 square miles of land eventually involved, at least three scenarios can be described for implementation and assessment monitoring.

First, employ a cadre of field and laboratory staff to periodically census the whole resource; this approach would be expensive and funded probably over a short term, if at all. Alternatively, self-monitoring and reporting by all landowners could be instituted. Self-monitoring is conducted by many States for point-source discharges, but may result in poorly implemented programs of questionable integrity (Chapter 10). Self-monitoring programs typically generate additional compliance monitoring because agencies would need to confirm or spot-check reports. Finally, a survey with sampling of selected sites could be started to infer results across the population. Whatever the choice, it will require close cooperation among many Federal and State agencies, as well as nongovernment organizations, district conservationists, and landowners.

The Research and Monitoring Committee of the Regional Ecosystem Office in Portland (REO) has the authority and provides the foundation for integrating Federal monitoring efforts in this region. Given the regional scale of the salmonid issues, the extensive Federal holdings in the region, current funding levels, and the previous leadership in monitoring protocols shown by Federal research laboratories, the Federal agencies appear to be a logical choice for coordinating this effort. However, it is essential that states, Tribal and other governmental parties be involved in developing the core monitoring strategy to ensure comprehensiveness and support for implementation. Once agreement is reached on a sampling design, indicators, and database management, there should be periodic reviews by, and consultations with, nonfederal technical staff from the agencies, as well as universities, industries, and environmental groups. This might best be accomplished through technical working groups such as described by Hayslip (1993).

Critical Agency concerns include what should be monitored and how (including by whom, where, and when), and whether individual and aggregate conservation plans are protecting and restoring salmonids. The where and when of monitoring are discussed under monitoring design in Section 15.2.3; the what of monitoring is outlined in Section 15.2.4, which focuses on indicators and sampling. Recommendations on whom should conduct monitoring in various instances are covered in the implementation portion of this document (Chapter 16).

15.1.4 Cooperative Support

A useful monitoring program needs the support of a computerized-database management system for timely data entry, storage, retrieval, analysis, and reporting. Such a system will be more responsive if it links Federal and nonfederal lands and draws support from both Federal and nonfederal institutions. Given the scope and complexity of the potential data, it is essential that data be converted quickly and accurately into relevant information (MSG 1993; Paulsen et al. 1991). Moreover, digital databases (including geographic information systems) ought to be easily retrievable by all interested parties.

Organizing a successful monitoring program of such coniplexity requires considerable Federal coordination and leadership. Such organization ensures that the collected data will have utility and

more knowledge than we have now will accrue concerning our effects on these systems.

Owners of private lands and managers of public lands will need to cooperate in conservation planning and monitoring activities because they will be the important users of the monitoring information. For example, biological integrity and sustainable levels of natural resource production (Figure 15-1, Paulsen and Linthurst 1994) are ecological and social goals that now concern to some degree the community of landowners, managers, and scientists. Common objectives can be attained with cooperative monitoring activities and practical ecosystem management. We recommend that a coordinated private-State-Federal monitoring and assessment program be implemented in the PNW on both Federal and nonfederal lands. Henjum et al. (1994) and McCullough and Espinosa (1996) have made similar calls for rigorous monitoring programs. The Research and Monitoring Committee for the President's Forest Plan is currently examining how to implement such a program on Federal lands; extending this effort to nonfederal lands in the Pacific Northwest would greatly enhance salmonid conservation planning.

15.2 Recommended Strategy for Monitoring Salmonid Conservation Activities

In the remainder of Chapter 15 we propose a strategy with eight activities for monitoring salmonid conservation. This monitoring strategy is based on the discussion of existing monitoring programs in Chapter 10 and the preceding general guidelines.

1. Develop a set of assessment questions or objectives that the monitoring should address. MacDonald et al. (1991) consider this the most critical step in monitoring. For example, determine the proportion of stream miles in the region (or a particular basin) that support summer salmonid populations (or salmonid spawning); determine the relationship of riparian buffer width (or condition) and various measures of stream condition (e.g., sedimentation, temperature, LWD, channel coniplexity); assess whether prohibited activities are occurring and with what frequency (e.g., harvest activities in riparian buffers).

2. Determine the indicators that will be used to assess biotic and abiotic conditions: ensure that these indicators can be related to the ecological values, the natural and anthropogenic stressors, or both. Include biological, habitat, and stressor indicators to assess biological condition, diagnose the site's environmental conditions, and evaluate the management and landscape conditions that affect the more proximal indicators (Karr and Dudley 1981; Karr et al. 1986; Messer 1990; Hughes et al. 1992; Fore et al. 1996). If hydrology and sediment transport are critical planning elements, implementation and assessment monitoring should include land-use types and extents within the watershed. If biodiversity is a concern, indicators should focus on ecosystem structure and function from the genetic to the landscape levels versus focusing on an indicator species (Noss 1990; Landres 1992; NRC 1992). If early detection of stress and recovery are concerns, changes in species composition of r-selected species and disappearance of sensitive species may be the most useful indicators (Schindler 1987). MacDonald et al. (1991), Rapport (1992), and Cairns et al. (1993) stated that good indicators are sensitive to multiple stressors and responsive to general disturbances yet have relatively low sampling error. They should also be easily measurable, interpretable, and cost-effective. In addition, useful indicators are biologically and socially relevant, anticipatory of future changes, and diagnostic of particular stressors. Such indicators are integrative of a number of different stressors. Hughes (1993) demonstrates how stream indicators were evaluated through use of these characteristics.

3. Use the index concept in selecting the sampling period, sampling sites (e.g., streams) and sampling locations at the sites, as well as in data analysis. Indexing is the process by which data collection and analysis are logically focused on particular times, places, and indices (Hughes et al. 1992).

- Index Period. Although aquatic systems change markedly with seasons, many variables generally look the same from year to year during the same season, unless perturbed. Thus aquatic systems can be sampled when they are 1) least varying, 2) most likely to be stressed by perturbations of concern, and 3) safely and economically sampled. This period (an index period) will be the summer or early fall for most Pacific Northwest streams, but may be other seasons if spawning or sediment loading are concerns (Plafkin et al. 1989).

- Index Sites. In the same manner, a subset of all stream miles or lakes can be sampled to avoid taking a census of them all (see the following guideline). These sites should include reaches on rivers as well as streams, especially when dealing with anadromous and potamodromous fishes. A set of these sample data can be statistically assembled to represent the total stream or lake population. (Note that here and in subsequent cases the term "population" is used in a

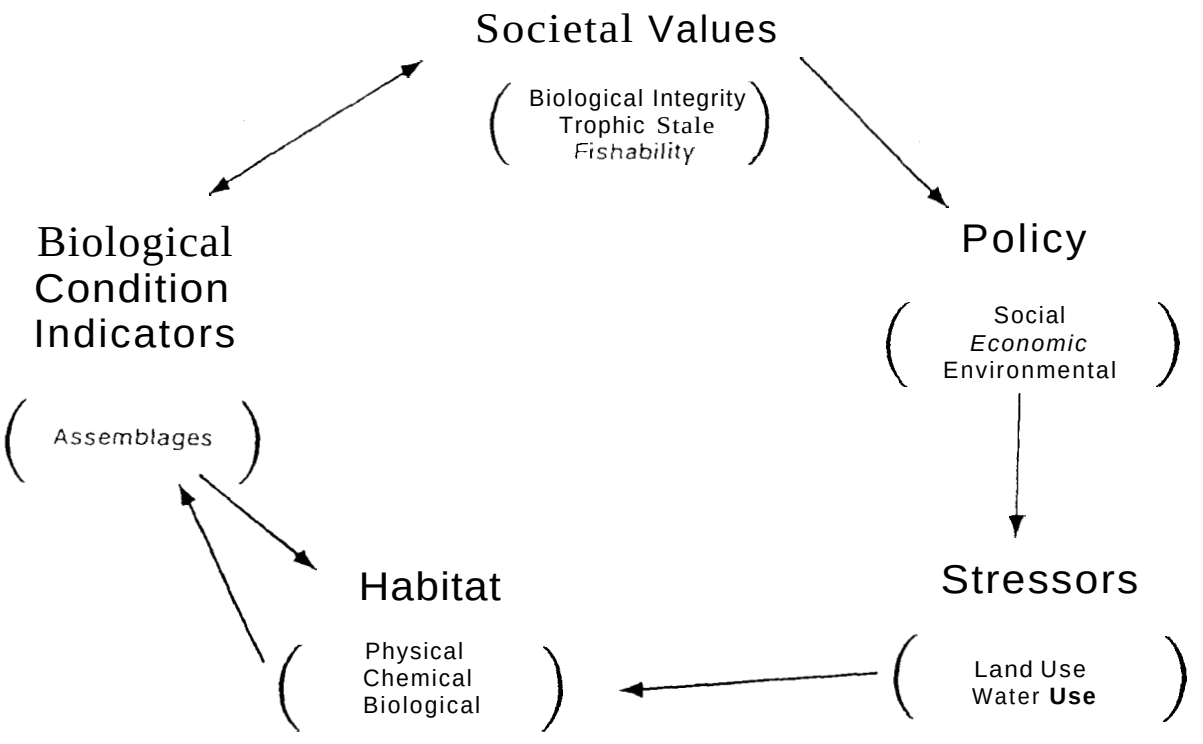


Figure 15-1. Relationships between societal values, policy, and stressor, abiotic condition (habitat), and biological condition indicators. Biological condition indicators are linked to societal values and biological condition assessments influence policy only through societal values, Also, biological condition indicators determine our choice of habitat indicators, which are the proximate determinants of biological condition. From Paulsen and Linthurst (1994)

statistical sense and refers to all stream miles or lakes of interest in the region.) This process has been very successful in assessing human opinions through political polls and market surveys, but has received remarkably little attention in ecological monitoring.

- Index Stations. At a single stream site, reach, lake, or watershed there are numerous macro- and microhabitat characteristics that could be evaluated. If a single sample inadequately captures the complexity of a site, as is the case with most biological and physical habitat indicators, it is useful and cost effective to index the site by randomized systematic samples of different variables. These may be composited by habitat type to represent the site. The rationale for this sanipling protocol is to assess the complexity of the site while limiting the cost of sampling and processing. On the other hand, because stream water is usually well mixed, a single index sample may suffice for estimating water quality for an entire reach.

- Numerical Indexes. The large amounts of data that may be generated from each site are often most useful if they can be converted or reduced into readily understood, summarized information. This is the role of numerical indexes. A numerical index, like a composite sample, synthesizes large amounts of information so that it can be easily displayed and understood. It is intended for nonspecialists more than for specialists, but it can offer considerable ecological insight when examined from the perspective of many sites through time. Examples are indexes of biological integrity (Karr et al. 1986; Kerans and Karr 1994; Fore et al. 1996) and *an* index of landscape stressors (Hughes et al. 1993). The

ecological merit of an index is a function of the quality and type of variables used to generate it (Barbour et al. 1995, Karr et al. 1986).

4. Develop a sampling design that is appropriate for answering assessment questions (item 1 above). If the ecosystem or region is the appropriate level for management or reporting, the monitoring and assessment also must be at these levels (Landres 1992; Paulsen et al. 1991; FS and BLM 1994a). Assessing whether HCP conditions are being met or if aquatic ecosystem conditions are improving or degrading across large regions requires data sampled at long temporal and large spatial scales (Hughes et al. 1992; Barrett et al. 1993; FS and BLM 1994a). Populations of sites, rather than individual sites, must be emphasized. Status-and-trend estimates must demonstrate a strong statistical foundation so uncertainty can be explicitly represented in confidence terms (Stevens 1994). These requirements support use of a probability sample of streams, which would be clearly representative and free of subjectivity in comparison with hand-picked sites. Populations should be stratified after sampling; this allows multiple interpretations of the data, permits detection of unanticipated issues, and improves precision if the misclassification rate of streams in various possible strata approaches 20% (Stevens 1994). Hall et al. (1978) also support an extensive poststratified design because it provides the greatest temporal and spatial perspective and takes the least time for evaluating condition and assessing cause-effect relationships. According to the REO (1995) and Botkin et al. (1994), the monitoring design should offer efficiencies of scale across large areas, distribute sites to reveal significant spatial variability, and include enough sites to determine statistical reliability.

5. Establish reference conditions (e.g., historical or natural, relatively undisturbed watersheds and stream segments) as standards against which conservation efforts may be measured. The goal of conservation need not be to achieve the reference condition. Frequently, the goal will be to reverse trends in resource condition so that they begin heading towards natural conditions. Because of the great diversity and multiple scales of the landscape and salmonid conservation issues, as well as variation in natural rates of disturbance, reference conditions will likely be derived from a variety of methods, including regional reference sites with minimal human disturbance, historical conditions, and models developed from such information (Platts et al. 1987; EPA 1990; Messer 1992; NRC 1992; Barrett et al. 1993; Hughes 1995). In general, naturalness can be estimated from the presettlement species complement, from the predicted degree that the system would change if humans were removed, or from the amount of cultural energy needed to retain the current system (Anderson 1991).

Natural disturbances of the landscape (e.g., fire, floods, drought, mass wasting) and variable oceanic and atmospheric conditions (El Niño, coastal upwelling intensity, climate) complicate the use of reference sites for establishing salmonid habitat or ecosystem standards. Even in undisturbed systems, streams may attain a variety of states in response to periodic disturbances and subsequent recovery (Reeves et al. 1995). Consequently, reference conditions should be defined to include the natural range of conditions occurring across the region or basin. Defining reference conditions to include the range of natural variation protects us from attempting to make all watersheds and rivers behave in the same manner. It also offers a disturbance gradient and spatio-temporal framework against which the extent of anthropogenic disturbances can be compared and with which the relationship between watershed conditions and salmonid responses can be modeled. This does not mean that because watersheds experience natural disturbances that human disturbances are insignificant. It simply provides a reference for evaluating the various degrees of the two sources of disturbance, as well as the conditions occurring in the absence of disturbance. Given the extent of human disturbance in the region, locating reference sites may be difficult with a probability sampling design so it will likely need to be amplified with subjectively chosen sites. A principal goal of the regional monitoring program outlined below is to produce a database from which the Agencies can better develop reference conditions for HCPs.

6. Apply the data in answering resource management questions, or in developing new assessment questions. Although this seems obvious, data are frequently collected but left unused. Certainly, data can be used to identify watersheds or stream sections where habitat has improved, remained the same, or degraded, and to determine the association of such changes with stressors. The focus here should be informational rather than punitive, assuming that management guidelines were followed. Monitoring information is also useful in assessing the successes and failures of the conservation program and validating or invalidating the principles incorporated in the planning efforts. These assessments will probably take decades for many issues. On the other hand, implementation monitoring can and should produce rapid alterations in land use if prohibited activities are violated, detected, and corrected.

Finally, the information is useful for demonstrating increased or decreased biological integrity and salmonid populations and comparing such changes to changes in habitat. Here again, tlie focus is more on research (e.g., validation of conservation principles) than compliance as long as approved conservation practices are implemented. The Watershed Analysis Coordination Team (WACT 1995) states that "existing data [are] adequate to accurately determine the current and historical status and distribution of aquatic biota" for low intensity analyses. However, current databases and distributional information for salmonids are spotty and based largely on presence information or subjective assessments. Data for other biota are even less reliable. Reliance on salmonids alone to assess biological integrity ignores many of the indicator concepts discussed above.

7. Evaluate the effectiveness of the strategy and its results. Using the results of monitoring, we recommend that the Agencies produce brief annual reports for public review and periodic research synthesis papers that are prepared by scientists trained in statistics and ecology. Regular program peer-and-participant reviews and recommendations for modifications are another essential part of monitoring. Because the implementation and assessment monitoring constitute the necessary data-acquisition phases for ecosystem research and management, evaluating the level of effort periodically becomes crucial: stable funding is needed to support the program with competent and dedicated staff.

8. Identify ecosystem elements and processes requiring additional research. Although this is not a major objective of monitoring on nonfederal lands, it is an activity that validates the inipleniented scientific and management practices; hence, it is recommended that suggestions for future research be part of a monitoring program and the issues identified be passed on to research institutions if not funded by the program.

15.2.1 Monitoring Implementation of HCPs and other Conservation Activities

HCPs and other salmonid conservation plans will probably contain a variety of provisions because of differences in current and attainable conditions among ecoregions and basins. We expect that virtually all HCPs prepared using this guidance will involve monitoring the implementation of land-use controls to reduce hydrological modifications, sediment transport, and riparian disturbance. Many will also require implementing activities that improve water quality and physical habitat structure. Some HCPs may involve removal of non-native fish species or introduction of beaver and LWD.

HCPs should include an approved and consistent implementation monitoring program; implementation monitoring is the process by which the Agencies determine if landowners are complying with provisions in their HCPs. To be most effective, baseline data should be collected before conservation activities begin. All data should be GIS compatible and entered into the database along with an indication of its sources (landowner, agency) to facilitate tracking of progress. Implementation monitoring is needed to ensure that prohibited activities do not occur and that permitted activities follow specific guidelines in the plan. Table 15-1 recommends a number of indicators potentially needed for an iniplenientation monitoring program. Not all indicators will need to be monitored in every instance, but others may need to be added to suit specific conditions and objectives. Most reconmended indicators are based on land use and land cover, resource extraction practices, pollution controls, and physical habitat structure. Indicators are discussed in more detail in Section 15.2.4.

15.2.2 Monitoring Effectiveness of HCPs and other Conservation Activities

The objectives of assessment monitoring will vary somewhat because of differences in land use and attainable conditions throughout the region, although there should be consistency in design and indicators to the greatest degree possible. If all HCPs involve monitoring to assess the effectiveness of their land-use controls, then ultimately we can determine the degree to which salmonids and their habitat have been restored or protected. To accomplish this goal, the focus of the monitoring should be on the aquatic and riparian ecosystems and include physical, chemical, and biological indicators. In addition, assessment of watershed conditions, which is a focus of implementation monitoring, will also provide information for adaptive management.

Because rapid detection of trends depends on early and precise assessments of condition, HCPs should encourage assessment monitoring that is started as soon as a consistent and rigorous program is developed by the Agencies. As with implementation monitoring, a large database will be produced requiring a large database-management system. These data will be useful for quantifying the relationships between various land uses and the response of salmonids and their habitats. Thus, both remote sensing and site visits are complementary in assessment monitoring. For example, to determine tlie extent and duration of riparian protection from farming, grazing, and logging, remote imagery aggregated over the drainage and site can be

Table 15-1. Recommended indicators for implementation monitoring.*

Assessed	Indicator
HCP description	Latitude, longitude, or UTM coordinates, area affected, initiation and completion dates [R]
Hydrology	% catchment with old uneven growth, closed and open canopy, nonforest, barren (forestland) [R] % catchment forested, shrubland, grassed, row-cropped (cropland) [R] % catchment grassland, shrubland, barren (rangeland) [R] % catchment forested, grassed, barren, impervious (urban, mining) [R] % stream miles channelized, ditched, piped (urban, cropland, mining) [R] %wetland [R] Wetland condition [S] Range condition [S] Water withdrawals [R,S]
Sediment transport	% catchment with mass wasting [R] % eroding stream banks (if specified in HCP) [R,S] Road density and proximity to streams [R] Harvest, roading, and restoration techniques [S] Tillage techniques [S] Mine site location and reclamation [R,S] Construction site sediment retention [R] Range condition [S]
Energy transfer and temperature	% channels with riparian forest within 10, 100, and 1000 m [R]
Water quality: nutrients and toxics	Lagoon capacity and integrity (confined livestock facilities, point source discharges, mines) [S] Effluent chemistry (point sources, irrigation return flows, storm drains) [S] Random, multispecies, whole effluent bioassays (point source discharges, mines, irrigation return flows) [S] Chemical applications [S] Irrigation techniques [S]
Physical habitat structure	% riparian zone within 100 m with natural riparian woody plants [R] % road crossings with inadequate culverts [S] % unscreened diversions [S] % impassable dams [R, S] Size, number, and location of LWD (if required in HCP) [S, R in large rivers] Frequency of off-channel habitats and LWD in riparian zone [R,S] Livestock density and timing [S] Livestock watering locations [S] Riparian fencing and forage condition [S]
Stream and riparian biota	Nonnative species and stocks (if required by HCP) [S] Beaver sign (if in HCP) [S] Fish stocking levels [A] Aquatic vertebrate species presence [S]

* A = Agency data, R = remote sensing, S = site inspection

employed; then, site visits are needed to assess the intensity of those land uses as well as their impact on aquatic life and physical and chemical habitat. Table 15-2 recommends a number of indicators for an assessment monitoring program; others will likely be added and some may be found inaccurate or imprecise in some ecoregions. As with implementation monitoring, we recommend several indicators representing each of the six monitoring categories listed in the table. Indicators are discussed in greater detail in Section 15.2.4.

15.2.3 Sampling Design for Monitoring Implementation and Assessment of HCPs and other Conservation Activities

Because of continued declines in widely ranging salmonids, a substantial proportion of the hundreds of thousands of stream miles and square miles of watersheds in the Pacific Northwest may eventually be covered under HCPs or other types of salmonid conservation agreements. To contain costs for agencies and landowners, technical innovations and training are employed. For example, a combination

Table 15-2. Recommended indicators for assessment monitoring.' Assumes indicators as described in Table 15-1 are provided.

Assessed	Indicator
Hydrology	Quantity and timing of peak and low flows (calibrated staff gage) Channel scour (scour chains) Discharge (measure) Valley type (map) [R]
Sediment transport	% fines (Wohlman pebble count at 100 intervals) Substrate size (Wohlman pebble count at 5 locations along 11 transects) % eroding bank (visual count at ends of 11 transects)
Energy transfer and temperature	% channel with riparian forest at 10, 100, and 1000 m [R] Extent and type of riparian vegetation in canopy, mid-layer, and ground cover (visually estimate classes at ends of 11 transects) % canopy cover (densiometer at ends of 11 transects) % channel and banks with anthropogenic disturbance (visual count at ends of 11 transects) Intensity of anthropogenic disturbances along channel and banks (visual count at ends of 11 transects) Extent of open channel with algal or macrophyte blooms (5 locations on 11 transects)
Water quality	Temperature (recording thermograph, summer low flow, 7-day, 0.5 hour recording frequency) Nutrients (N & P forms, lab analysis) Dissolved oxygen (recording thermograph, summer low flow, 7-day, 0.5 hour recording frequency) Turbidity & chloride (lab analysis) Toxics (whole-fish tissue contamination; for mines, point sources and irrigation return flows only; focus on suspected heavy metals and organics) Conductivity (meter) Intergravel dissolved oxygen (subset of sites only where FPOM is prevalent; stratified random sample of egg pockets during incubation, syringe sampling)
Physical habitat structure	Channel sinuosity and aspect (bearing compass at centers of 11 transects) Off-channel habitats (visual and measurements at ends of 11 transects) Residual pool volume (thalweg profile: depth measurements at 100 intervals along entire site) Channel cross section dimensions (measure width and depth at 5 points along each of 11 transects) Substrate size and complexity (Wohlman pebble count and % fines < 1 mm at 5 points along each of 11 transects) Bank undercutting, height, erosion, slope (measure with clinometer and **rod** at ends of 11 transects) LWD (record size, placement in bankful channel, number of pieces via running tally) Cover (include off-channel pools, undercut banks, LWD, overhanging and instream vegetation at 5 points along each of 11 transects) Gradient (clinometer, at centers of 11 transects) Riparian vegetation structure (species composition, DBH measurements across transects or plots within riparian zone of influence)
Stream and riparian biota	Microbial respiration (only where toxics and organic enrichment expected; sediment dissolved oxygen consumption with field respirometer) Periphyton (enriched streams only; quantitative sample from 11 transects; species composition and abundance) Benthic macroinvertebrates (quantitative samples from 11 transects; species composition and abundance) Fish and amphibians (systematic sample of a reach length that is 40–50 times the wetted stream width) Riparian birds (only for multispecies HCPs; systematic sample of 1 km reach at 11 sites during breeding season; species composition and abundance)

* R = remote sensing, all others require site inspection.

of remote sensing, using both aerial photography and satellite imagery, with site visits for selected indicators would facilitate moniitoring. The use of such technology could save hundreds of work years and millions of taxpayer dollars. Similarly, data gathered by trained field staff following a sampling design are more reliable, less costly, and less disruptive for the landowners. These examples reveal again the necessity for cooperation.

Initially, one might logically focus monitoring on an individual drainage or set of reaches covered in an HCP. Conimonly, the drainage or reaches are subjectively selected to be representative or typical, but rarely is the assumption that such sites are representative tested through statistical evaluation. Subjective site selection is a common approach for persons concerned with a particular place. However, because of the variability in streams in the Pacific Northwest one must either census all stream reaches in an HCP or region, or have a large sample size. A rule of thumb in survey sampling designs advises a minimum of 20–50 sites to adequately represent a population. If that population contains streams of markedly different sizes, gradients, and substrates, a sufficient number of sites is needed so that each class contains 20–50 sites. Note that this does not mean 20-50 samples are needed within each reach.

A regional sample survey or census is also iniportant for placing individual conservation activities into an ecoregional and basin context. Because of widespread deterioration in salmonid habitats, it will frequently be necessary to establish reference conditions from information acquired from outside the area covered by the HCP. Reference conditions are essential for determining desired directions and outcomes for restoration, for setting quantitative criteria for evaluating progress, and for assessing the effectiveness of the HCP. Minimally disturbed reference sites offer a means for determining if trends in assessed variables result from the effects of the HCP or from changes in climate, passage, harvest, and hatcheries. Although establishment of reference conditions is desirable, reference sites are likely to be scarce or absent from extensively disturbed regions. In these instances, reference conditions may be established by other means, including historical data, quantitative models, and professional judgement (Hughes 1995).

Even if few HCPs are implemented for nonfederal lands, it will be useful to determine regional conditions and trends via a regional sample survey or census. Both landowners and the Agencies will need to know whether various watersheds and reaches in HCPs are in markedly better or worse condition than others in the basin and ecoregion. Such information is also useful for developing planning elements of the initial HCP. An ecosystem approach to salmonid conservation involves tailoring management prescriptions to the specific capacities of particular systems. Unfortunately, as the preceding sections have shown, we often lack the information to develop ecoregional standards, let alone watershed- or site-specific standards. Regional-scale monitoring can provide data for establishing these standards before conservation activities in such places are developed.

Another argument for regional HCP monitoring is that salmonid conservation and biodiversity are fundamentally regional issues. Whether conservation planning becomes commonplace, the responsible State and Federal agencies are beginning to recognize that their current assessment and compliance monitoring programs are inadequate. The various programs have differing assessment questions, indicators, reference standards, and database management systems. Consequently, their sampling designs, sampling methods, and reported results appear contradictory or biased. **A** good deal of the responsibility for the "salmon problem" rests with the management agencies responsible for the salmon and their habitats, and the inadequacies of their monitoring and reporting. Correcting these shortcomings requires developing a more rigorous and consistent regional monitoring program.

We recommend a multi-State regional sample survey for several reasons. 1) There are ecoregional patterns in biotic and abiotic factors at both multi-State and basin scales (Hughes et al. 1994) and it takes a regional approach to assess them. 2) Summarizing segment level information in an organized manner facilitates making landscape-level statements (Conquest et al. 1994; Yoder and Rankin 1995). Landscape-level statements are needed for regionally distributed organisms like salmon. 3) It would be prohibitively expensive to inventory or census all nonfederal lands and stream miles in the region with the quantitative indicators needed to accurately and precisely assess status and trends. 4) Regional assessments of status and trends should be conducted in a statistically consistent and unbiased manner to instill public confidence and to avoid not identifying a problem when one exists (Type II error; Rhodes et al 1994). *5)* Fragmentary monitoring fosters fragmentary ecosystem management and social systems (Karr 1994). **6)** Previous emphasis on site- and watershed-specific assessments is a key reason that it took so long to determine the regional extent of deteriorating salmon stocks, although many would argue that signs have been evident for decades. 7) **A** multi-state and multi-agency survey elevates monitoring to a regional concern and makes results less easy to ignore.

There are several advantages of a randomized sample survey over other sampling designs. If we infer aquatic conditions from nonrandomly picked sites, we cannot estimate the uncertainty of our assessments or the biases of our inferences (Larsen et al. 1995). A randomized sample survey is necessary to determine population characteristics (Larsen et al. 1994) and to allow unbiased condition and trend estimates. A randomized survey sample also allows data assessment by basin, ecoregion, political unit, ownership class, or any other regional phenomenon.

We propose that the Agencies adopt something like EPA's EMAP sampling design. The EMAP design is easily intensified (Serveiss 1995; Stevens 1994) if detailed information is needed for a single HCP or basin, yet it offers great cost savings by not requiring intensive inventorying of entire drainages. In addition, the EMAP design facilitates accurate and precise inference about resources throughout the region of concern, something that the currently popular stream inventories or subjectively chosen fixed sites cannot offer. Equally important, EMAP's randomized design and nionitoring frequency offer rapid assessment of regional status and trends, which would be exceedingly costly or time consuming via an inventory approach.

EMAP pilots support using several sampling design features. 1) Use a randomized grid (e.g., Overton et al. 1990) to select approximately 200 stream points from digitized versions of GS 1:100,000 scale topographic maps. 2) Use a classification or weighting process to ensure that all channels of interest are represented and that the smallest and most numerous ones are not over represented. 3) Check maps and conduct field visits to ensure that the streams are target systems. 4) Determine ownership and obtain access permission. 5) Record reasons for non-targets and inaccessibility, and draw a replacement from the stream population.

For implementation monitoring, it may be possible to census all watersheds with HCPs or other conservation agreements as long as the nuniber or areal extent remains small. As the areas in conservation agreements increase, a randomized sample of watersheds should be obtained through use of a grid design (Stevens 1994). These are sampled without replacement to ensure that the maximum number of watersheds are eventually monitored. Digitized watershed boundaries are overlain by classified land-use and land-cover data, such as that from thematic mapper with a 30 meter pixel size, from 1:40,000 scale color infrared air photos, or both. Site visits should include focused inspections at points of particular concern (e.g. feed lots, treatment facilities, extraction practices) as well as random inspections of extensive activities (stream crossings, riparian fencing, road construction). By maximizing the sample size, this sampling design is oriented towards assessing condition and it facilitates assessments of subpopulations of waters.

For assessment monitoring, representative samples may be obtained by sampling a stream section equivalent *to* 40 channel widths long that is centered on the stream point designated by the computer; locations can be confirmed with maps and a GPS unit. At the site, we recommend using a randomized systematic sample design to collect quantitative data on physical, chemical, and biological variables at multiple stations. We also suggest compositing of multiple biological samples for each assemblage by major habitat type, although there remains disagreement in the scientific community regarding the benefits of compositing samples (see e.g., Fore et al. 1996). During the index period(s) of interest, sampling variances (temporal, crew, measurement) can be evaluated by resampling 10–15 randomly selected sites. Land use and land cover within watersheds should be assessed via remote sensing data as described above. For the following three years, repeat this process at approximately 200 new stream points selected each year, including the resampling. In year five, resample all sites sampled in year one, in year six all those sampled in year two, and so on. This sampling design balances our ability to assess status, the precision of which is increased by increased sample size, and to detect trends, the sensitivity of which is improved by sampling the same waters annually (Larsen et al. 1995). More detailed sampling designs than are possible to develop herein must be developed by the Agencies following consultation with other nonfederal and Federal partners.

15.2.4 Physical, Chemical, and Biological Indicators

Quantitative indicators like those proposed by EMAP (Hughes 1993) and McCullough and Espinosa (1996) are needed to ensure that ecological signals are discriminated from spatial, temporal, and methodological variances, thereby aiding rapid detection of trends and accurate estimates of status.

A set of variables is recommended to measure the iniplenientation and effects of conservation activities on the attributes and processes identified as critical (Tables 15-1 & 15-2). These variables include several representing each of the major planning elements discussed in Chapter 14. In addition, they were chosen to assess how well the conservation activities produce the desired changes in physical and chemical habitat. Not all variables need to be monitored in all situations; instead appropriate variables depend on the type of impact and conservation efforts proposed (see parenthetical comments in Tables 15-1 and 15-2). A set of

biological variables is included because the biota are the fundamental indicators of concern. Also, monitoring landscape, habitat and biota will enable the Agencies to validate whether the activities are having the desired effects. Linkages between major planning elements and the recommended indicators should facilitate adaptive management and HCP modifications when results are contrary *to* expectations.

Based on EMAP pilots (Hayslip et al. 199.5; Klemm and Lazorchak 1995), such data could be collected with about 6000 work hours per year (3 persons × 10 hours per site × 200 sites per summer). A similar number of hours is needed annually for geographers to conduct the land-use and land-cover investigations. Additional resources would be required for air photos, gear, travel, and sample processing. Additional indicators (riparian birds, salmonid spawning, salmonid genetics) or additional sites would add to the costs. A substantial investment is also needed for database management, data analysis, and reporting, but these additional costs are common to all current monitoring programs. This could be a very cost-effective investment compared with the cost of current Federal programs (e.g., EMAP, NAWQA).

The following discussion supports assessment of particular indicators or indicator groups, All are commonly nionitored by various institutions, though not in the same ways. In pilot variance studies in various parts of the country and in the hands of specialists, they have all been found to be precise and responsive *to* stressors, especially when data are composited and metrics are integrated into multimetric or multivariate indices, Although field methods have been widely tested (Baker et al. 1994; Hayslip et al. 199.5; Klemm and Lazorchak 1995), the results of these studies are mostly in preparation for submission to journals. The data demonstrate that different indicators respond differently to different stressors, revealing the need for multiple indicators of different types. **At** a minimum, we recommend consistent, quantitative monitoring of the landscape, physical and chemical habitat variables, benthic niacroinvertebrates, and aquatic vertebrates at all assessment monitoring sites. In addition, we reconmend monitoring microbial respiration in urban and mining streams; periphyton in agricultural and rangeland streams; and riparian birds and salmon genetics, spawning, and rearing in random subsets of streams. Monitoring multiple indicators at as many sites as possible is recommended.

Stressors. Human uses of the landscape and riparian zone in large part govern the condition of the water body. This information is available from digitized land-use and land-cover data for each watershed and available remote imagery for each drainage (Rhodes, et al. 1994; Paulsen, et al. 1991). It is used to assess the type, condition, and extent of woody riparian vegetation, both for the site and for a random sample of upstream stream sections. Types, intensity, and extent of watershed, basin, or regional land use and land cover are used to estimate areal disturbance, road density, stream crossings, stream proximity, and migration barriers. Fish stocking and harvest rates, livestock stocking rates, water withdrawals, and historical information about resource exploitation are also useful, but often more difficult to acquire. We recommend monitoring land use, land cover, and historical and present resource extraction rates through use of remote and print data. In addition, site inspections are needed for ground truthing and for the indicators listed in Table 15-1, which vary with land use.

Physical Habitat Structure. There is considerable agreement among State and Federal agencies in the need to monitor many structural components of streams and riparian ecosystems. Riparian indicators include valley type, riparian vegetation structure, human disturbance, and canopy cover. Channel indicators include sinuosity, aspect, gradient, bank erosion or channel incision, bank height, bank undercutting, thalweg profile, depth, and width. An additional set of indicators of habitat complexity include large woody debris, fish cover, and a number of substrate variables (size, complexity, percentage of fines). Because they typically determine the basic capacity and current character of the site, we recommend quantitative measurements of these indicators wherever possible. Current research on the relationships between these variables and fish populations promise to make them even more useful in the future.

Water Qualify. The chemical condition of the water offers a useful means to classify streams by their mineral type and nutrient status, and water quality is a powerful signal for landscape scale stressors. Highly mobile indicators like chloride and nutrients are among the first signals of landscape level perturbations and they are useful measures of landscape revegetation and nutrient retention. Although water chemistry may be of less importance in many forested watersheds, it is critical where land uses include human settlements, agriculture, and livestock grazing. At a minimum, we recommend monitoring temperature, nutrients, turbidity, conductivity, chloride, and dissolved oxygen. If salmonid spawning is of concern, then intergravel dissolved oxygen should also be monitored at a subset of sites during the incubation season. Continuous monitoring of dissolved oxygen and temperature may be recommended in areas likely to experience reduced concentration or supersaturation during late summer. In the vicinity of mines and

point sources, monitoring for toxics contained in fish tissue is advised.

Microbial Respiration. This is an assemblage and community-level measure of the reach's carbon processing rate that can be rapidly and inexpensively evaluated on site with simple techniques. It is especially sensitive to the amount of fine particulate organic matter in the system, as well as to the presence of toxics that are likely to result from past or present mining and urban activities. Because most carbon processing in streams results from microbes, this is a useful assemblage to evaluate. It also offers a direct, quantitative measure of a biological ecosystem function. We recommend monitoring microbial respiration for distinguishing chronically toxic or organically enriched waters from a population of streams.

Periphyton. Periphyton assemblages are key primary producers in stream ecosystems, and streams with high rates of primary production are typically our most productive salmonid waters. On the other hand, excessive levels of periphyton signal nutrient enrichment. Composition of periphyton assemblages is also an excellent indicator of low-level or chronic sedimentation in streams, which is determined from the relative abundances of motile and nonmotile diatoms. We recommend monitoring periphyton where fish are absent or in regions where nutrient enrichment is likely.

Benthic Macroinvertebrates. Benthic assemblages are a popular and easily monitored set of stream organisms. They occupy a key position between the algal and detrital food base and fish. Also they are species rich and numerous enough to occupy a large array of habitats and niches. This diversity in structure and function facilitates their use in assessing the effects of numerous perturbations, from water quality changes to sedimentation. Benthos, like periphyton, are especially useful for assessing aquatic biological integrity in fishless waters. They should be monitored in all streams.

Aquatic Vertebrates. Aquatic vertebrates are typically the top carnivores in PNW streams. Forested streams support 0–5 fish species and 0–3 amphibian species, all of which can be effectively sampled with the same gear and methods. Although headwater streams occasionally contain no vertebrates, most larger coastal streams support 2–3 lamprey species, 2–3 sculpin species, and at lower elevations and inland 1–3 minnow species, in addition to salmonids. Each species provides information about the biological integrity of the reach and each is susceptible *to* different types of anthropogenic stressors. Protocols focusing only on salmonids miss key information about other anadromous species and resident vertebrates. Aquatic vertebrate monitoring is recommended for all streams.

Salmon Spawning and Rearing. A primary concern of this document is restoration and protection of salmon populations in the PNW. The most appropriate methods to determine achievement of that objective is to monitor spawner abundance or redds and smolt production. The former can be accomplished by aerial surveys, traps, or stream walks, depending on stream size, and the latter is best monitored through use of outmigrant traps. In cooperation with the State fishery agencies, a randomized subset of streams should be monitored for salmon rearing and spawning.

Riparian Birds. Birds provide an easily sampled indicator of how a terrestrial vertebrate assemblage responds to riparian conditions as well as to conditions in the stream and watershed. They are best sampled by competent ornithologists during the breeding season, when populations are most stable and censuses easily taken through sightings and songs. Birds are of great interest to the public and are monitored by Federal agencies through the Partners in Flight Program, the National Breeding Birds Survey, and Christmas bird counts. We recommend monitoring birds wherever multi-species HCPs are developed and at a subset of sites as an indicator of riparian integrity.

Several steps facilitate data collection, analysis, and reporting. At the site, portable data recorders or standardized field sheets facilitate data entry in the database management system. Verification and validation checks on the data are needed for quality control. Measurement data converted to numerical indicators are useful for comparing resample variance with population variance. By running exploratory analyses (scatter plots, principal components analysis, regression analysis, correlations) indicator patterns and behaviors can be easily assessed. To express status and track trends, we recommend selecting ecologically meaningful indicators that possess relatively little sampling variance but considerable responsiveness to stressors. Such indicators should be plotted as histograms, cumulative frequency distributions, maps, or pie graphs for interpretation by interested persons. These indicators are also used to demonstrate regional patterns, temporal patterns, and proportions of the stream population that exceed or fail to meet various criteria. Criteria and reference conditions should be developed from regional reference sites, historical information, models, and expert judgement (Rhodes et al. 1994; Hughes 1995). The ecological acceptability of conditions and trends is a value judgement (Kay and Schneider 1992), but marked changes from reference conditions can be considered marginal or severely impaired (Barbour et al. 1995; Hughes 1995). A regional landscape-aquatic

database also facilitates examination of the effects of natural variability among ecoregions and basins as well as the effects of floods, droughts, fire, and ocean conditions.

15.2.5 Other Monitoring Issues

A similar sampling and analysis design is recommended if the Agencies choose to sample only at the watershed scale. In such cases the stream population is necessarily smaller and less variable, so a smaller random sample can be obtained. For statistical consistency and reliability, a general rule is a minimum of 30 sites for each class of interest. Because of greater site proximity, there may be advantages in dropping some indicators so that crews can sample more than one site per day. Note, however, that it only takes seven watershed-specific monitoring programs to result in the 200 sites recommended for monitoring in a regional program, but without the regional applicability. Clearly, if regional information is desired it is most cost effective to design a regional survey as proposed. If the interest is both regional and watershed-specific, then the grid can simply be intensified where needed to provide both. Whatever the landscape scale, a probability sampling design is essential for reliably assessing status and trends, unless the entire resource can be rapidly and quantitatively censused.

Regardless of whether the monitoring is conducted at the watershed or regional scale most indicators and monitoring protocols should be the same so that results can be integrated. It is essential to include stressor information in implementation and assessment monitoring. Riparian and instream biological, physical, and chemical indicators are needed if the monitoring objectives involve ecological assessment.

Streams and watersheds that are found to be in very good condition or highly productive of salmonids deserve protection as reference sites, biological refugia, sources of high quality water, or locations for studying natural process rates. Conservation activities that connect these refugia with others are more likely to be successful in protecting and restoring salmonids and biodiversity in general.

Several programmatic concerns should be incorporated into an effective monitoring program. All monitoring personnel must receive consistent training and repeat sampling should be conducted at a subset of locations by other persons to ensure among-watershed comparability and to assess sampling variance. To evaluate ecoregional and basin patterns, watershed-scale data must be aggregated to the larger spatial scales. This requires common indicators among watersheds and Agency coordination to analyze and integrate the data. Regional results should be regularly reported through workshops, the media, and informational brochures.

A successful monitoring program depends on adequate, long-term funding. Contributions from both Federal and nonfederal sources, including landowners and the general public, might consist of money, staff, or equipment. We encourage the maximum amount of cooperation possible in the collection, analysis, and reporting of the data because of the great value of salmonid ecosystems and the importance of high quality information for making rational decisions about natural resources. Two issues, identifying funding sources for long-term monitoring and designating a lead agency (or agencies) for database management and reporting, need to be resolved if salmonid conservation activities are to be accurately evaluated and if monitoring information is to be used effectively both to adaptively manage and to guide future planning efforts.

In addition to the monitoring discussed above, other forms of monitoring and assessment are desirable. 1) There is a substantial need for rigorous stock assessment through use of genetic and morphometric analyses of salmonids in all sub-basins of the Pacific Northwest; this data will aid in delineating ESUs and addressing biodiversity issues. Also, it could be coupled with the other forms of fish monitoring. 2) We need to assess salmonid diseases within basins and at distribution breaks. Disease is a poorly studied limiting factor, and information on disease may also assist in defining ESUs. 3) In addition to those in Oregon, aquatic diversity areas and spawning "hot spots" should be located in the other States and in other regions of Oregon. These areas serve as foci for protection and restoration, and they are useful for setting recovery expectations for disturbed sites. 4) Continued monitoring of adults and smolts is needed at dams and hatcheries, especially the effects of these perturbations on the timing and abundance of salmonid migrations. As dams are removed and hatchery practices modified, pre- and post-modification monitoring will provide useful information on their effects. 5) Monitoring of salmon harvest is needed to document the successes and failures of the various options. 6) A central fish database of historical information is needed. Such a database was developed from museum data for Oregon (Hughes et al. 1987), but it needs amplification with other forms of less rigorous data on fish species and abundances (e.g., collections without museum specimens, probable distributions). To our knowledge, the other States in the region lack even the museum database, despite the importance of knowing the fish species to expect in any watershed of the region, as well as the range and probable abundance of a species. Clearly, the monitoring of these more regional components of the salmonid

environment requires integration with the monitoring program discussed in the preceding subsections,

15.3 Summary

Considerable information exists about the successes and failures of compliance monitoring and the benefits and shortcomings of various assessment monitoring designs (reviewed in Chapter 10). In addition, there are sufficient examples of the advantages of quantitative indicators and indicators of biological condition, in particular. These are all only briefly examined in this volume.

To evaluate compliance and assess the effectiveness of management practices at both local and regional scales, a comprehensive interagency, Federal-nonfederal monitoring program is strongly recommended. It should be entered into only after a thorough examination of long-term objectives and goals. We need rigorous sampling designs and quantitative data (from physical, chemical, and biological indicators) to make informed decisions about those goals and objectives and to practice adaptive management in a rational manner. Unfortunately, if we continue to develop ecological and economic policies without monitoring strategies to measure their effects, it is likely that we will ultimately guarantee ecological and economic failure. Perhaps these shortcomings partially explain why Karr (1995), McCullough and Espinosa (1996), and Henjum et al. (1996) feel our current regulatory agencies do not respond to degradation in an effective and timely manner. We have the potential and the tools to do much better; we recommend committing the monitoring and management resources necessary *to* do so.

16 Implementation Strategy

"My grandfather taught me how to mend net, to close a gaping hole one knot at a time, measuring each mesh Fishing was all small acts, practiced with the devotion of the religious. .We owe following generations no less than restoration of the Northwest's wild salmon runs. Ours must De small acts practiced with the same devotion of my grandfather mending web "

James B. Petit
Solid Faith in Small Acts *in Illahee*, 1994

Conservation and restoration of salmonid habitat in the Pacific Northwest will require that a series of small acts be integrated into a comprehensive program. A successful conservation program aims to restore the natural function of landscapes—at least sufficiently to restore the processes and habitats supporting Salmonids. Each action can be viewed in the context of how well it protects or enhances salmonids and their habitats. Similarly, the conservation activities of individual landowners can be made most effective if they are woven into a regional habitat and salmon conservation program.

An implementation strategy involves the large temporal and spatial scale issues in salmonid conservation. In preceding sections of this document we have presented niuch information about why, where, and how we need to restore and protect salmonids and their habitats. We have also recommended why, where, and how monitoring should be implemented. In this chapter we discuss who needs to do what and when they need to do it. In Section 16.1 we recommend how a regional conservation plan should be developed. Section 16.2 discusses the implementation of site and regional monitoring programs. Finally, Section 16.3 summarizes additional issues that need to be considered when implementing a successful conservation strategy.

16.1 Development of HCPs and a Regional Salmon Conservation Strategy

The "what" portion of an implementation strategy includes developing HCPs and other conservation practices, monitoring the implementation and effects of those practices, and developing and evaluating the overall conservation program. Clearly, it is the responsibility of landowners and land managers, with Agency guidance, to develop conservation plans at the site or catchment scale. To date, HCPs for salmon have been developed primarily by large corporations, municipalities, or Federal agencies. If more salmonid stocks are listed under ESA, the need will increase to develop conservation plans for watersheds with multiple ownerships, including watersheds with many small private landowners that share ownership, as well as watersheds with mixed private, State, Federal, and Tribal ownership. In these instances, conservation planning becomes increasingly complex. Federal-nonfederal land exchanges could facilitate planning and land management in some of these cases, especially within the checkerboard ownership patterns that resulted from the last century's railroad lands. The scientific principles guiding conservation of aquatic and riparian habitats should not differ between Federal and nonfederal lands; however, conservation standards may be more conservative on Federal lands. Because of the desired size of the planning units, 20–200 square mile watersheds, we expect that most HCPs will eventually involve multiple landowners. The strategy in such cases is likely to take one of two courses. Where there are dominant or codominant owners, we recommend that they take the lead in HCP preparation, with contributions from fellow landowners proportionate to ownership. Where ownership patterns are more heterogeneous, watershed councils or cooperatives can be formed to either produce a plan via existing county or municipal staff or contract for one, as many of the large landowners do now. A growing number of watershed councils appear to be following this tack.

A regional plan or program is similarly problematic, but it involves a much larger spatial scale (region versus watershed). The Federal land management agencies through the Record of Decision (ROD) for the President's Forest Plan (FS and BLM 1994c) have established a program for Forest Service and Bureau of Land Management lands west of the Cascades. PACFISH (FS and BLM 1994a) proposes interim planning guidelines for anadromous salmonids on Federal lands east of the Cascades and INFISH (FS 1996) provides similar guidelines for

resident salmonids on Federal lands in the interior portion of the Northwest. *An Ecosystem Approach to Salinoizid Conservation* covers nonfederal, salnionid-bearing streams throughout major portions of California, Idaho, Oregon, and Washington. All of these reports document programs that would be enhanced if they were linked with one another and with other Federal, State, and Tribal entities into a comprehensive regional salmonid conservation program. Such a program should include listed species, at risk but currently unlisted species, and their ecosystems. In other words, the Agencies should work to ensure that this program and individual conservation plans incorporate an overall conservation strategy for Federal, as well as nonfederal, lands in tlie four-State region.

A strategy to implement a salmon conservation program on nonfederal lands requires a new perspective for Federal fisheries agencies. Many private landowners have demonstrated a willingness to change management practices. For example, many improvements in agricultural practices, range management, forestry management, and mining have occurred in the past 50–100 years, but these changes occurred slowly. As innovations proved effective and profitable, they were disseminated through the affected community. The Natural Resources Conservation Service (NCRS), formerly the Soil Conservation Service (SCS), practices a method that serves as one model for resource conservation planning on nonfederal lands. It is based on mutually identifying and discussing issues, identifying options or guidelines, and providing individual landowners sufficient information to change their actions voluntarily. The EPA practices an alternative model. EPA develops science-based criteria and best treatment practices that States can accept or modify with sufficient scientific support. The States list desired uses for water bodies and apply the necessary criteria to protect those uses. EPA and the States regulate tlie dischargers through permits, monitoring, and if necessary, fines. This process has been successful at markedly reducing point source pollution by private and public dischargers over the past 25 years. Both approaches involve substantial Federal investments either in field staff in the case of tlie SCS or in matching funds for municipal waste treatment in tlie case of EPA.

We recommend an implementation strategy that combines the best of both models. It should be science based and include a regulatory component because of the urgent need to change current practices in order to restore salmonids. In addition, there should be sufficient field staff to aid landowners in developing and implementing HCPs because of the fundamental philosophical and technical changes needed. Because salmonids are resources of national concern, Federal aid for planning, implementation, and monitoring is merited.

16.2 Monitoring Conservation Efforts Locally and Regionally

As discussed in Chapter 15, conservation planning must be monitored at both the site and regional levels. In addition to implementation and assessment monitoring, tlie process of developing HCPs must also be monitored to assure quality and regional consistency. But who should do the monitoring, and who should pay for it? Chapter 10 discusses some of tlie failures of well known monitoring programs. Given concerns with losing wild salmonids from the Pacific Northwest for their own sake, as well as for the enormous economic consequences that entails, we must strive to do a much better job of monitoring and enforcement.

16.2.1 Program Monitoring

At the level of program implementation and development of HCPs and other conservation agreements, we recommend two different activities. First, the HCPs themselves should be largely reviewed by Agency staff. We recommend at least bi-agency review if the HCP waters do not drain contiguous Federal land. **If** they do drain contiguous Federal land, the appropriate Federal land management agency should review the HCP and the Agencies should review the Federal conservation plan. These reviews should be conducted by staff that are professionally trained in the disciplines of geology, hydrology, soil science, aquatic ecology, riparian forest ecology, fisheries ecology, and where necessary, toxicology and engineering. Consistent plans should be the goal, at least within ecoregions and regardless of ownership. In addition, HCPs should complement existing State conservation programs, and the most complete analyses should be considered adequate for both purposes. Second, public comment, as required by Section 10 of ESA, should be requested before HCP approval. The comments and recommendations, along with the Agency responses, should be entered and tracked in the computer database. We also recommend that the overall conservation program itself undergo periodic peer review, similar to those already conducted by tlie Science Advisory Board for EPA, with reviewers representing other agencies, academia, and the private sector. This is common practice for ensuring the integrity of science and will add to the credibility of tlie Agency's conservation efforts.

16.2.2 HCP Implementation Monitoring

Implementation monitoring should mostly be conducted by Agency staff. This monitoring could be accomplished by contract or by State employees, with results entered directly into the Agencies central database. Although some State agencies have a better record than the Federal government in enforcing laid-use and pollution laws, this creates interstate inconsistency, an unnecessary additional layer of bureaucracy, and added overhead costs. In addition, an HCP is a contract with the Agencies, not with a State, although States may be integrally involved in other conservation efforts. Ideally, persons conducting the HCP reviews will also perform some of the implementation monitoring, especially site inspections. Because of the array of expertise involved, site visits are expected to require more than one staff member. The remote sensing portion of the implementation reviews require geographers and landscape ecologists with skills in GIS analysis and airphoto interpretation, but they too should be expected to conduct some ground truthing. Results and comments should be entered into the centralized computer database.

16.2.3 HCP and Regional Assessment Monitoring

Development of a regional assessment monitoring system for salmonid ecosystems is clearly an Agency responsibility, although if properly coordinated, it could include other Federal, State, Tribal, and private crew members. The same is true of crews monitoring HCPs, although in this case private landowners are likely to show more interest in the monitoring. Whatever the scale of the monitoring and whomever the employer, the crews should be trained consistently, use the same sampling methods and quality assurance protocols, and be dispersed randomly in the region to the greatest degree possible. Along with the assurance of repeat sampling by different crews, this will minimize biased results. If State crews are prohibited by their employers from crossing State lines to sample, if other Federal crews cannot sample on nonfederal lands, or if private crews can only sample on their own lands, mixed crews are not recommended. As with the other two types of monitoring, data should be stored in the centralized computer database.

16.3 Additional Issues in Implementing a Salmon Conservation Strategy

As discussed above in the nionitoring subsections, there is a clear need for a cooperative Federal, State, and Tribal effort in developing the computer database as well as support for the necessary database managers, computer programmers, and statisticians to ensure effective and responsive operation. The Agencies and others will also need to support a substantial database containing digitized remote sensing data as well as a periodically updated library of topographic maps and air photos, together with a librarian to coordinate them. Much of this information will also be useful to other persons interested in developing future HCPs.

Although the proposed program incorporates some of the highest technology and planning in landscape ecology and conservation biology, the results of the planning and monitoring should not be reported only to scientists. The public must remain actively engaged in this process. This may be best accomplished by preparing annual reports in an easily read and understood format for public consumption.

The databases, including the HCP implementation and monitoring results, should be used to improve our ability to develop ecoregion-specific management programs. A critical aspect of this entire strategy is to position ourselves to learn from and correct past and future mistakes. We recommend that conservation measures be reviewed and revised if monitoring or new research suggests inadequacies; the frequency of review would depend on specific concerns or issues, but adaptive management requires strong linkages between monitoring and the modification of conservation strategies.

We also see three issues relating to equitable treatment of landowners that are likely to be concerns in successful conservation planning. 1) Landowners that have previously managed their lands to conserve ecological integrity or biological diversity may be expected to restrict future resource exploitation more than those who have intensively and extensively exploited resources, particularly if these lands contain habitats critical to the persistence of salmonid stocks. This also is an issue in comparing forestland restrictions with urban, agricultural, and rangeland restrictions. No single land use should shoulder a disproportionate share of the burden in restoring salmonid habitats. We have attempted to make consistent recommendations for all land uses but suspect situations will arise where comparable restrictions are impractical. Alternative conservation trade-offs, land exchanges, tax breaks, or other incentives may provide means for rewarding good stewardship. 2) In contrast, multi-owner conservation efforts will occasionally include individual landowners that have been poor stewards and whose actions may limit salmon and ecosystem recovery throughout a planning area. Strategies (e.g., education, removal of Federal subsidies) will be needed to deal with these instances. 3) Violations of the antidegradation clause of the Clean Water Act, as well as listed species, occur on Federal lands. This is

another reason we recommend implementation and assessment monitoring of both Federal and nonfederal lands.

Salmon conservation and the enforcement of ESA come at a time when choices are increasingly limited. Marsh (1965[1864]) predicted this very condition for Pacific salmon over 100 years ago. Now we have the choice of driving more stocks and species toward threatened and endangered status or of managing our lands to avoid that situation. Given some present trends in the Pacific Northwest, salmonids not currently listed as threatened or endangered under the Endangered Species Act are at risk of listing in the future. Conservation efforts are far more likely to succeed if we conserve remaining relatively healthy salmonids—rather than drive them to listing and then attempt to restore them. For this reason, we strongly recommend development of HCPs or other forms of conservation planning and monitoring throughout the region, regardless of whether a particular stock or species is listed. Agency staff are mandated by other laws than ESA (e.g., Clean Water Act, National Environmental Policy Act), as well as by their knowledge of conservation biology, to manage proactively and to seek to prevent species from becoming threatened or endangered.

We also have two broader strategic concerns. First, a focus on the immediate physical and chemical habitat of salmonids and the land uses that affect them is insufficient to conserve salmon. Other activities, including hatchery and harvest practices, as well as water withdrawals and the operation of dams, need to be included in a successful conservation strategy. Even if conservation efforts eventually restore habitats throughout the region, salmonids may still decline unless we modify hatchery operations, promote terminal instead of mixed-stock fisheries, limit water withdrawals from thermally and low-flow stressed rivers, and modify or remove dams that impede upstream and downstream migration. Second, continuation of current growth rates in human population and resource consumption in the Pacific Northwest indicate that demands for resources—and the incumbent effects on salmonids—are likely to intensify. Ultimately, these root causes of environmental deterioration will need to be addressed as part of our conservation planning efforts.

These recommendations acknowledge that ecosystem management will be accomplished through many individual and independent actions. But they also acknowledge that if ecosystem management and salmon conservation are to succeed, each independent action must be integrated into a comprehensive program with a regional conservation objective. The science underlying landscape management and salmonid conservation is constantly changing; thus, implementing an effective strategy requires adapting to new information as it is developed. It is our belief that the planning elements contained in this document provide a foundation from which to build a successful strategy by applying what we already know about ecosystem function, as well as facilitating the collection of information that will allow us to improve planning efforts in the future.

Appendix: Information Sources

A.l Introduction

This appendix supports Part II, "Planning Elements and Monitoring Strategies," which itself builds on Part I, "Technical Foundation" of *An Ecosystem Approach to Salmonid Conservation*. Within Section A.2, we identify selected sources of information that may assist resource managers, regulators, and landowners in obtaining the necessary background data to develop comprehensive habitat conservation plans or to critically evaluate such plans. We have not compiled an exhaustive list of all sources, but we have provided examples of sources that individuals may use directly or may use as a guide for seeking other sources to meet their particular data needs. For each type of data, information is listed in a tabular format: tlie data coverage (i.e., regional or state) appears in the left column with the form of available data (e.g., maps, documents, information, GIS coverages, and other electronic databases); the data source appears to the right, along with a brief description of the data *(in italics)*, if available. Following this description is an alphabetical list of all addresses and Internet addresses of sources included in this appendix.

Users are cautioned about the changeability of electronic access to information. Because electronic information changes frequently, access to tlie World Wide Web universal resource locators (URLs), email addresses, and telephone area codes with numbers below may have changed since publication of this document. Consequently, as many access points as possible via many media have been provided. One strategy to recapture access is to enter an address using one less segment of it in tlie hopes of entering a server at one or two levels of specificity above the location of the sought information.

In Section A.3 we provide brief sketches of relevant Federal and State laws and regulations that may be of concern or use to landowners involved in conservation planning for salmonids and their habitats. This is not an exhaustive list, but we sought relevant information in the areas of land use, forestry, agriculture and pesticides, range, mining, water quality, instream flows, and channel alteration for the four States in the Pacific Northwest. Persons needing further information are encouraged to visit a university law library or appropriate web sites. Section A.4 presents a list of mailing addresses, phone numbers, FAX numbers, and internet addresses for various Federal and State government offices in the Pacific Northwest.

A.2 Regional Versus State-Specific Data and Sources

Data or data sources that apply to more than one of the States of California, Idaho, Oregon, and Washington are listed first under "Regional." If there are any State-specific sources (e.g., State offices of Federal agencies) or exclusively State-specific data (e.g., State of Washington Watershed Analysis Units Map), those entries follow. State-specific data that are available from one source for multiple States are included under "Regional," as are sources that apply generally to all locations regardless of the specificity of the data itself. Just because there are no "State-specific" entries does not mean there are no State-specific data from a source that also has data for other States in the region (and is thus listed under "Regional").

Data sources by category are illustrated in Table A. Data may be available in different forms (e.g., GIS layers, maps, digital, print) and sources may use different criteria for determining land or regional characteristics (e.g., Bailey's ecoregions versus Omernik's ecoregions). In addition, the same data may be available in various forms from different sources (e.g., 1:250,000 hydrology maps are available in hard copy from the Geological Survey's Map Distribution Center or in digital form from the Geological Survey's Node of the Geospatial Data Clearinghouse on the Internet). Finally, some data may be available from multiple vendors (e.g., hard-copy GS 7.5 minute topographic maps). In some cases, sources are addresses (postal or Internet) of sources that distribute data. In other cases, sources are citations to literature; documents must be obtained from a library or the publisher. Some data are identified with an asterisk (*) indicating that data are known to exist, but a specific data source, data availability (whether it is published or is available for public release), or the content of the data (whether the data source contains the data type) is not known.

Users are cautioned to evaluate the applicability of any data source relative to their specific needs.

This appendix is not nieant to be a comprehensive list of all available sources; other sources of applicable data may be available. The user may identify these when beginning with the provided examples and starting points. It is likely that once some of these sources of information are contacted by a user with data needs for a specific location or application, other, more specific data forms will be suggested. It is highly recommended that the user consult with identified local and State agencies and offices or local field offices of Federal agencies prior to acquiring data; less expensive or more site-specific data and information may be available at the local or State level given a specific data requirement for a specific location.

This appendix presents one table for each of fifteen types of data that may be useful for developing and evaluating habitat conservation plans: 1) Ecoregion, 2) Hydrologic Unit, *3)* Topography, 4) Geology, 5) Soils, 6) Current Vegetation, 7) Historical Vegetation, 8) Aerial Photographs, 9) Land Use History, 10) Precipitation, 11) Streamflow, 12) Stream and Surface Water Type, 13) Water Quality, 14) Fish Species Distribution, and 14) Threatened and Endangered Species.

Table A. Data source overview.

Data Type	States			
	CA	ID	OR	WA
1. Ecoregion	R	R	R,S	R
2. Hydrologic unit	R,S	R,S	R,S	R,S
3. Topography (and aspect*)	R	R	R	R
4. Geology	R	R	R	R
5. Soils	R,S	R	R	R,S
6. Vegetation (current)	R,S	R,S	R,S	R,S
7. Vegetation (historical)	R	R	R,S	R
8. Aerial photographs (current)	R,S	R,S	R,S	R,S
9. Land-use (or environmental†) history	R	R	R,S	R
10. Precipitation	R	R	R,S	R,S
11. Streamflow	R,S	R	R	R,S
12. Stream and surface water type	S	R‡	S	S
13. Water quality	R,S	R,S	R,S	R,S
14. Species distribution—fish	R,S	S	R,S	R
15. Endangered/threatened species	R,S	R,S	R,S	R

* Aspect can be derived from topographic map data.
† Environmental history may be derived from land-use history and historical vegetation.
‡ Surface water type for Idaho may be available from regional sources under "Streamflow."
R Regional. State affiliates or offices of Federal agencies or data sources are included with regional sources.
S State-specific.

Table A-I. Sources for ecoregion data

Data coverage	Data form	Source
Regional	Map	Bailey, R. G. 1978. Description of the ecoregions of the United States. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, 324 25th St., Ogden, UT 84401-2310, (801) 625-5437. Developed initially to provide a spatial *framework* for the *U.S.* Department of the Interior, Fish and Wildlife Service (FWS), National Wetlands Inventory.
	Map	FS (Forest Service). 1993. National hierarchical framework of ecological units, ECOMAP. U.S. Department of Agriculture, Forest Service, Washington, D.C. Developed to provide a consistent framework for *the* implementation of ecosystem management by the Forest Service at the national, regional, and forest planning levels. The map units are differentiated by multiple factors including climate, physiography, geology, soils, water, and potential natural communities.
	Map/GIS Digital	Omernik, J. M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77:118–125. Electronic copy available (no fee) on the Intemet (World Wide Web) through the Geological Survey's node of the National Geospatial Data Clearinghouse as an *ARC/INFO* export tile: *http://nsdi.usgs.gov/nsdi/wais/water/ecoregion.html*. Initially developed to classify streams for water resource management, derived from those factors considered most important in controlling water quality in a given area, including land surface form, land use, soils, and natural vegetation.
	Map	Thiele, S. A., C. W. Kilsgaard, and J. M. Omernik. 1993. The subdivision of the Coast Range Ecoregion of Oregon and Washington. On file at U.S. Environmental Protection Agency, 200 SW 35th St., Corvallis, OR 97331.
State		
Oregon	Maps	Clarke, S. E., D. White, and A. L. Schaedel. 1991. Oregon, USA, ecological regions and subregions for water quality management. Environmental Management 15:847–856. Bryce, S. A, and S. E. Clark. 1996. Landscape-level ecological regions: linking state-level ecoregion frameworks with stream habitat classifications. Environmental Management 20:297–311.

Table A-2. Sources for hydrologic unit data.

Data coverage	Data form	Source
Regional	Map	GS (Geological Survey). 1994a. Hydrology map of the 48 conterminous United States. Map scale 1:250,000. U.S. Department of the Interior, Geological Survey, Washington, D.C. This map is available (no fee) on the Internet (World Wide Web [WWW]) through the GS node of the National Geospatial Data Clearinghouse, http://nsdi.usgs.gov/nsdi/wais/water/huc25O.html, as an ARC/INFO export file. See also GS-Info./Maps below. GS (Geological Survey). 1994b. Hydrologic unit map of the United States. Map scale 1:7,500,000. U.S. Department of the Interior, Geological Survey, Washington, D.C. *Hydrologic units are watersheds defined by topographic drainage divides.*
State		
California	Map	GS (Geological Survey). 1978. Hydrologic unit map, 1978, State of California. Map scale 1:500,000. U.S. Department of the Interior, Geological Survey, Reston, Virginia.
Idaho	Map	GS (Geological Survey). 1982. Hydrologic unit map, 1981, State of Idaho. Map scale 1:500,000. U.S. Department of the Interior, Geological Survey, Reston, Virginia.
Oregon	Map	GS (Geological Survey). 1976. Hydrologic unit map, 1974, State of Oregon. Map scale 1:500,000. U.S. Department of the Interior, Geological Survey, Reston, Virginia.
Washington	Map	GS (Geological Survey). 1976. Hydrologic unit map, 1974, State of California. Map scale 1:500,000. U.S. Department of the Interior, Geological Survey, Reston, Virginia.
	Map	WDNR (Washington Department of Natural Resources). N.y. Watershed Analysis Unit (WAU) map. Map scale 1:100,000. Washington Department of Natural Resources, Photo & Map Sales, P.O. Box 47013, Olympia, WA 98504-7013, (360) 902-1234.

Table A-3. Sources for topography data.

Data coverage	Data form	Source
Regional	Maps	U.S. Department of the Interior, Geological Survey, Map Distribution Section, Map Sales, Federal Center, Box 25286, Denver, CO 80225, (303) 236-7477. *The Geological Survey produces 1:24,000 (7.5 minute) topographic quadrangles for all areas within the conterminous United States. Topographic maps may be ordered from the Map Distribution Section at the above address or from local vendors; the Map Distribution Section can provide a list of local and regional vendors.*
	GIS-Info./ Maps	U.S. Department of the Interior, Geological Survey, Public Inquiries Office (PIO), Building 3, Room 3128, Mail Stop 522, 345 Middlefield Road, Menlo Park, CA 94025, (415) 329-4390. *PIO assists the public in selecting and ordering of all GS products and provides counter service for GS topographic, geologic, and water-resources maps and reports. The Menlo Park office covers the States of California, Idaho, Oregon, and Washington.*
	Maps/GIS Digital	GS node of the National Geospatial Data Clearinghouse, WWW address at http://nsdi.usgs.gov/nsdi/. *Metadata that describe geology, water, and mapping sets available on the World Wide Web. Many data sets are available on-line, including digital elevation model, land use/land cover, and water resources. An interactive search can be performed using States or latitude and longitude of a specific location. The program will search for all GS spatial data available for that location.*

Table A-4. Sources for geology data.

Data coverage	Data form	Source
Regional	Info/maps	U.S. Department of the Interior, Geological Survey, Geological Inquiries Group (GIG), 907 National Center, Reston, VA 22092, (703) 648-4383. GIG provides information and answers inquiries concerning all aspects of geology, such as the geology of specific areas, energy and mineral resources, earthquakes, volcanos, geochemistry, geophysics, and geologic map coverage. GIG produces Geologic Map Indexes, by State. Geologic maps show the underlying geology of a specific area and often include other information such as the presence of rock outcrops, unstable soils (sometimes determined by bumpy, uneven ground surface using aerial photographs), etc.
	GS-info /maps	U.S. Department of the Interior, Geological Survey, Public Inquiries Office (PIO), Building 3, Room 3128, Mail Stop 522, 345 Middlefield Road, Menlo Park, CA 94025, (415) 329-4390. PIO assists the public in the selection and ordering of all GS products, and provides counter service for GS topographic, geologic, and water-resources maps and reports. The Menlo Park office covers the States of California, Idaho, Oregon, and Washington).
	Maps/ GIS Digital	GS node of the National Geospatial Data Clearinghouse. WWW address at http://nsdi.usgs.gov/nsdi/. Metadata that describe geology, wafer, and mapping sets are available on the World Wide Web. Many data sets are available on-line, including digital elevation model, land use/land cover, and water resources.

Table A-5. Sources for soils data.

Data coverage	Data form	Source
Regional	Soil surveys (books)	U.S. Department of Agriculture, National Resource Conservation Service (NRCS) (formerly Soil Conservation Service [SCS]), West Regional Office, Sacramento, CA. Additional soil information for specific States or counties can be obtained through the NRCS State Conservationist or Resource Inventory Specialist at the appropriate NRCS State office, or through local NRCS field offices. Addresses and phone numbers for State offices are as follows: California, 2121-C 2nd Street, Davis, CA 95616, (916) 757-8200; Idaho, 3244 Elder Street, Room 124, Boise, ID 83705-4711, (208) 378-5700; Oregon, 1220 S. W. Third Avenue, Room 1640, Portland, OR 97204-2881, (503) 414-3028; Washington, Rock Pointe Tower II, Suite 450 W. 316 Boone Avenue, Spokane, WA 99201-2348, (509) 353-2337. Provides Soil Surveys in book form by County, primarily covering agricultural areas, which include aerial photographs and soil maps. Soil type descriptions include slope, permeability, and other useful information, Contact the Regional or State office and request the phone number of the field office for your County. Field offices generally distribute the Soil Surveys for that County free of charge. Inquire as to whether the survey covers your area of interest before requesting the survey book itself. Staff may also assist you in determining the soil type for your location.
	Database	National Soil Characterization Database. The database of the Soil Survey Laboratory (SSL), National Soil Survey Center, currently contains analytical data for more than 20,000 pedons of U.S. soils. The data are available on one standard CD-ROM disk. To obtain technical information about these data, contact Steven L. Baird, USDA, Natural Resources Conservation Service, National Soil Survey Center, Soil Survey Laboratory, Federal Building, Room 152, MS 41, 100 Centennial Mall North, Lincoln, NE 68508-3866, (402) 437-5363. To order the data, contact the USDA, Natural Resources Conservation Service, National Cartography and Geospatial Center, 501 Felix St., Bldg. 23 (P.O. Mail 6567), Fort Worth, TX 76115, (800) 672-5559. Current Price: $50.00 for single CD-ROM disk. The National Soil Characterization Database is composed of the SSL working computer files. It includes data that may or may not represent the central concept of a soil series or map unit and pedons sampled to bracket a range of soil properties within a series or a landscape; all such data are retained in the database. Users unfamiliar with a given soil may want to consult a knowledgeable soil scientist to determine how well the data represents a soil series; the database has not been edited to remove all of the erroneous or sometimes misleading data. Users are responsible for assessing the accuracy and applicability of the data. The characterization data are stored in a fixed length, column positional, tab-delimited file structure, with a two-record freeform header, in master and State data sets, ASCII format.

Table A-5. Sources for soils data.

Data coverage	Data form	Source
	GIS Digital	USDA Natural Resources Conservation Service (NRCS), National Cartography and Geospatial Center (NCGC), Fort Worth, TX 76115-0567, (800) 672-5559. For information on the availability of GIS coverages of soil information for specific States, contact the NRCS State Conservationist or Resource Inventory Specialist at the appropriate NRCS State office. (Addresses and phone numbers are listed under "Soil surveys" above.) *The NRCS maintains three soil spatial (GIS) databases including the Soil Survey Geographic Data Base (SSURGO), the State Soil Geographic Data Base (STATSGO), and the National Soil Geographic Data Base (NATSGO). Components of map units in each database are generally phases of soil series. Information such as particle size distribution, bulk density, available water capacity, soil reaction, salinity, and organic matter is included for each major layer in the soil profile. Also included are data on flooding, water table, bedrock, subsidence characteristics of the soil, and interpretations for erosion potential, engineering, building and recreational development, and cropland, woodland, wildlife habitat, and rangeland management.* *SSURGO provides the most detailed level of information and is used primarily for farm and ranch conservation planning; range and timber management; and county and parish, township, and watershed resource planning and management. These data are digitized from original soil survey maps (see Soil Survey Reports above). Data are collected and archived in 7.5 minute topographic quadrangle units (scale 1:15,840 to 1:31,680). Digital coverage for many areas in the United States is currently limited.* *STATSGO is used primarily for river basin, State, and multicounty resource planning. Soil maps for STATSGO are made by generalizing more detailed soil survey maps or, where detailed maps are not available, by integrating data on geology, topography, vegetation, and climate, as well as Landsat images. The GS 1:250,000-scale topographic quadrangle series is used as a map base; data are collected and archived in one degree by two degree topographic quadrangles.* *NATSGO is used primarily for national, regional, and multistate resource appraisal, planning, and monitoring. The NATSGO map was digitized at a scale of 1:7,500,000 and is distributed as a single data unit for the conterminous United States coverage.* *The NCGC operates both a Geographic Resource Analysis Support System (GRASS) and an ARC/INFO GIS. Other formats may be available.*

Table A-5. Sources for soils data.

Data coverage	Data form	Source
	Database inventory	The National Resources Inventory (NRI) is a database form of the inventory of land cover and use, soil erosion, prime farmland, wetlands, and other natural resource characteristics on nonfederal rural land in the United States. For more information on data-collection methods and results for specific States or regions, contact the NRCS State Conservationist or Resource Inventory Specialist at the appropriate NRCS State office. (Addresses and phone numbers are listed under "Soil Surveys" above.) Data can be ordered off the World Wide Web at http://www.ncg.nrcs.usda.govlnri.html. To obtain the data analysis software or for additional information on the NRI program, contact USDA-Natural Resources Conservation Service, Resources Inventory and Geographic Information Systems Division, P.O. Box 2890, Washington, D.C. 20013, (202) 720-4530. To order data write the USDA-Natural Resources Conservation Service, National Cartography and Geospatial Center, Fort Worth Federal Center, Bldg. 23, Room 60, P.O. Box 6567, Fort Worth, TX 76115-0567, (800) 672-5559. Inventories are conducted at 5-year intervals by the U.S. Department of Agriculture's Natural Resources Conservation Service (NRCS), formerly the Soil Conservation Service (SCS), to determine the conditions and trends in the use of soil, water, and related resources nationwide and statewide. The NRI is linked to NRCS's extensive Soil Interpretations Records database to provide additional soils information. Data elements consistent within the NRI database among the last three (1982, 1987, and 1992) NRIs: I) farmstead, urban, and built-up areas; 2) streams less than 1/8 mile wide and water bodies less than 40 acres; 3) type of land ownership; 4) soils information—soil classification, soil properties, and soil interpretations such as prime farmland; 5) land cover/use—cropland, pasture land, rangeland, forest land, barren land, rural land, urban and built-up areas; 6) copping history; 7) irrigation—type and source of water; 8) erosion data—wind and water; 9) wetlands—classification of wetlands and deep-water habitats in the U.S. (1982 and 1992 only); IO) conservation practices and treatment needed; 11) potential conversion to cropland. The 1992 NRI is the most extensive inventory yet conducted, covering 800,000 sample sites, representing the Nation's nonfederal land—some 75% of the Nation's land area. Data collected in the 1992 NRI provide a basis for analysis of 5- and IO-year trends in resource conditions. New data elements added for the 1992 NRI include 1) streams greater than 1/8 mile wide and water bodies by kind and size greater than 40 acres; 2) Conservation Reserve Program land under contract: type of earth cover—crop, tree, shrub, grass-herbaceous, barren, artificial, water; 3) forest type group; 4) primary and secondary use of land and water; 5) wildlife habitat diversity; 6) irrigation water delivery system; 7) Food Security Act (FSA) wetland classification. For a more detailed understanding of the data element characteristics, request a copy of the "Instructions for Collecting 1992 NRI Sample Data." Many data items in the 1992 NRI are consistent with previous inventories.

Table A-5. Sources for soils data.

Data coverage	Data form	Source
		Database Availability: The NRI database is available to the public on four CD-ROMs (ISO 9660 format) at $50 per disk. Each disk contains data for a collection of States that form a contiguous region (CD #1 includes Arizona, California, Colorado, Hawaii, Idaho, Montana, Nevada, New Mexico, Oregon, Utah, Washington, Wyoming). Each disk includes separate files containing the Soils Interpretations Records and spatial data sets for mapping NRI data (see "Spatial Data Sets," below). All files are flat ASCII files. Data can be downloaded on a State-by-State basis if disk storage space is limited. Database documentation is provided. Spatial Data Sets: Spatial data sets of boundaries of Major Land Resource Areas, 8-digit hydrologic units, and counties are provided on each CD for the region corresponding to the NRI data with and without Federal lands. These data sets contain the same spatial identifiers used in the NRI database allowing NRI users to create interpretive maps. The data are provided in Geological Survey, DLG-3 formatted files on the four data CDs for the appropriate region. GRASS-GIS vector formatted files are included on the data analysis software. Documentation on spatial databases is provided.
State		
California	Info/ Database	National Resource Conservation Service (NRCS): USDA, Natural Resources Conservation Service, 2121-C Second Street, Suite 102, Davis, CA 95616, (916) 757-8262. NRI Information via email: nri@ca.nrcs.usda.gov. In California, over 8,000 sample sites were used. Nonfederal land represents 94% of the State's land base. At each sample point, information is available for three years: 1982, 1987, and 1992. From this time series, changes and trends in land use and resource characteristics can be estimated and analyzed for a IO-year period.
	GIS Digital	Natural Resources Conservation Service (NRCS) SSURGO and STATSGO spatial databases. See entry under "Regional" above.
Idaho	GIS Digital	Natural Resources Conservation Service (NRCS) SSURGO and STATSGO spatial databases. See entry under "Regional" above.
Oregon	GIS Digital	Natural Resources Conservation Service (NRCS) SSURGO and STATSGO spatial databases. See entry under "Regional" above.
Washington	Map	Soil Erosion Potential Map for WAU, Washington Department of Natural Resources, Photo & Map Sales, P.O. Box 47013, Olympia, WA 98504-7013, (360) 902-1234.
	GIS Digital	Natural Resources Conservation Service (NRCS) SSURGO and STATSGO spatial databases. See entry under "Regional" above.

Table A-6. Sources for data on current vegetation.

Data coverage	Data form	Source
Regional	Land-cover maps	U.S. Department of the Interior, Geological Survey, National Cartographic Information Center Cover (NCIC), Western Mapping Center, 345 Middlefield Road, Menlo Park, CA 94025, **(415)** 328-4309. NCIC offices accept orders for aerial photographs and satellite imagery and sell custom cartographic products such as GS digital data, color and feature map separates, orthophotoquads, and land-use and land-cover maps.
	Maps/GIS digital	Geological Survey (GS) node of the National Geospatial Data Clearinghouse at http://nsdi.usgs.gov/nsdi/. Metadata that describe geology, water, and mapping sets are available on the World Wide Web. Many data sets are available on-line, including digital elevation model, land *use/land* cover, and water resources. The Land Use and Land Cover (LULC) data files describe the vegetation, water, natural surface, and cultural features on the land surface. GS provides these data sets and associated maps as a part *of* its National Mapping Program.
	Database inventory	The National Resources Inventory (NRI) is an inventory of land cover and use, soil erosion, prime farmland, wetlands, and other natural resource characteristics on nonfederal rural land in the United States. See NRI annotafions in Table ***A-5*** for description and source.
	Database survey, biological	National Biological Service (NBS), Gap Analysis Program (GAP). These data are meant to be used at a scale of 1:100,000 or smaller (such as 1:250,000 or 1:500,000) to assess the conservation status of vertebrate species and vegetation cover types over large geographic regions. The data may or may not have been assessed for statistical accuracy. Data evaluation and improvement are ongoing. NBS makes no claim as to the data's suitability for other purposes. Contact Michael D. Jennings, Coordinator, GAP Analysis Project, National Biological Service, Idaho Cooperative Fish and Wildlife Research Unit, University of Idaho, Moscow, Idaho. GAP Data Sets are available on-line on the Internet via the World Wide Web at http:l/www.nr.usu.edulgaplimaplpreus.html. See also: NBS homepage at http:l/www.its.nbs.govlnbs and the National Biological Information Infrastructure, Directory of Biological Data and Information at http:llwww.its.nbs.gov/nbiildirectory/html.

Table A-6. Sources for data on current vegetation

Data coverage	*Data form*	*Source*
		NBS is part of the USGS, and it began GAP to map species diversity and identify priority areas containing species currently not represented in areas managed for their natural values. The basic GAP data layers are 1) land cover, 2) vertebrate species distributions, 3) land ownership, and 4) land management. Vegetation maps are constructed from Landsat imagery. For each species of terrestrial vertebrates in an area, a habitat association model is used to identify polygons on the vegetation map considered suitable habitat. Known occurrences of each species are compiled by county from published literature and museum records. *Range maps for the species are then estimated as those polygons with vegetational classes considered suitable habitat that occur in counties with known species occurrence. GAP is primarily organized at the State level, as a cooperative effort of the NBS with other public and private organizations. This and other data may be available through the Information Transfer Center, Mr. Rich Gregory, Director, 1201 Oak Ridge Drive, Suite 200, Fort Collins, CO 80525, (970) 226-9401 or (970) 223-9709, FAX (970) 226-9455 or the Technology Transfer Center, Mr. Phil Wondra, Director, P.O. Box 25287, Denver, CO 80225-0287, (303) 969-2590, FAX (303) 969-2160, email Phil_Wondra@nbs.gov. These offices form the core of the Information and Technology Services branch of the National Biological Service at http://www.its. nbs.gov.* *The Information Transfer Center responds to the information needs of the natural resource community in three major ways: by providing references to the scientific literature on a topic of interest; by functioning as a research liaison between scientists and anyone needing assistance with a challenging natural resource problem; and by editing, publishing, and distributing manuscripts written by NBS scientists. A critical information transfer tool is the compilation of references to the scientific literature. This service is the responsibility of the Bibliographic Information Branch of the Information Transfer Center. More than 300,000 citations are contained in the databases, indexed and distributed as Wildlife Review and Fisheries Review. Some 24,000 citations are added to the database each year.* *Wildlife Review and Fisheries Review is available by subscription through the U.S. Government Printing Office. The databases are also available in CD-ROM format from a private vendor. Libraries and other information sources now have the capability of conveniently conducting their own literature searches using the databases on CD-ROM. The Biological Assistance Branch provides technical assistance to agency personnel throughout the United States.* *See also Table A-8, Aerial Photographs (current).*
State		

Table A-6. Sources for data on current vegetation.

Data coverage	Data form	Source
California	Maps	NCIC State Affiliate: Map and Imagery Laboratory Library, University of California, Santa Barbara, CA 93106, (805) 961-2779.
	Data sets	National Biological Service (NBS), Gap Analysis Program (GAP) See entry under "Regional" above. GAP Data Sets are available on-line on the Internet via the World Wide Web at http://www.nr.usu.edu/gap/imap/preus.html for the following California ecoregions: Cascade Ecoregion, Central West Ecoregion, East Sierra Nevada Ecoregion, Great Valley Ecoregion, Modoc Plateau Ecoregion, Mojave Ecoregion, North West Ecoregion, Sierra Nevada Ecoregion, Sonoran Ecoregion, and South West Ecoregion.
Idaho	Info	NCIC State Affiliate: Idaho State Historical Library, 610 N. Julia Davis Dr., Boise, ID 83702, (208) 334-3356.
	Data sets	National Biological Service (NBS), Gap Analysis Program (GAP). See entry under "Regional" above. GAP Data Sets are available on-line for the State of Idaho on the Internet via the World Wide Web at http://www.nr.usu.edu/gap/imap/preus.html.
Oregon	Info	NCIC State Affiliate: Oregon State Library, Public Services, Salem, OR 97310, (503) 378-4368 .
	Data sets	National Biological Service (NBS), Gap Analysis Program (GAP). See entry under "Regional" above. Oregon GAP Vegetation GIS coverage. In Oregon, the NBS, through the Idaho Cooperative Wildlife Unit, is working with the Oregon Department of Fish and Wildlife, The Nature Conservancy, and Defenders of Wildlife. GAP Data Sets are available on-line on the Internet via the World Wide Web http://www.nr.usu.edu/gap/imap/preus.html for the State of Oregon.
Washington	Info	NCIC State Affiliate: Washington State Library, Information Services Division, Olympia, WA 98504, (360) 753-4027.
	Map	Forest stand age map in IO-year increments; hydrologic maturity map, Washington Department of Natural Resources, Photo & Map Sales, P.O. Box 47013, Olympia, WA 98504-7013, (360) 902-1234.
	Data sets	National Biological Service (NBS), Gap Analysis Program (GAP). See entry under "Regional" above. GAP Data Sets are available on-line for the State of Washington on the Internet via the World Wide Web at http://www.nr.usu.edu/gap/imap/preus.html.

Table A-7. Sources for data on historical vegetation.

Data coverage	Data form	Source
Regional	Aerial photographs	U.S. Department of the Interior, Geological Survey, Photographic Library (LIB-P), Mail Stop 914, Building 20, Box 25046, Federal Center, Denver, CO 80225, (303) 236-1010. *The Photographic Library contains a collection of over 250,000 photographs (predominantly black-and-white) taken during GS studies. A few photographs taken before the founding of the GS (1879) are included in the collection. Library staff will prepare lists of selected photographs in response to specific requests. Photographs are indexed by subject and location.*
	Scientific literature	Franklin, J. F., and C. T. Dyrness 1973. Natural vegetation of Oregon and Washington. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Oregon State University Press, Corvallis, Oregon. *This book describes the natural vegetation communities (plant associations) in the Pacific Northwest, and could be used to identify the indigenous plant communities likely to have inhabited a given general area. Consult your State Library or State University Library for availability. It is no longer printed by the Federal government.* *Shultz, S. T. 1990. The Northwest coast: a natural history. Timber Press, Portland, Oregon. **Data are known to exist but a specific source for the data is not known, the availability is not known, or the content of the data is not known.*
State		
Oregon	Map	*Historical maps from OSU Map Library, Valley Library, Oregon State University, Corvallis, OR 97331, (541) 737-3331.
	Map	* Historic vegetation maps of Oregon counties, digitized by FS.
	Scientific literature	Teensma, P. D., J. T. Rienstra, and M. A. Yeiter. 1991. Preliminary reconstruction and analysis of change in forest stand age classes of the Oregon Coast Range from 1850 to 1940. Technical Note OR-9. U.S. Department of the Interior, Bureau of Land Management, 825 NE Multnomah Street; P.O. Box 2965; Portland, OR 97208 (503) 231-6274.

Table A-8. Sources for aerial photographs

Data coverage	Data form	Source
Regional	Photographs	WAC Corporation (Aerial Photography), 520 Conger Street, Eugene, OR 97402-2795, (800) 845-8088 or (541) 342-5169. *WAC can provide coverage of western Oregon, western Washington, and northern California.*
	Maps/GIS digital	U.S. Department of the Interior, Geological Survey, National Cartographic Information Center (NCIC), Western Mapping Center, 345 Middlefield Road, Menlo Park, CA 94025, (415) 328-4309. *NCIC offices accept orders for aerial photographs and satellite imagery, and sell custom cartographic products such as GIS digital data, color and feature map separates, orthophotoquads, and land-use and land-cover maps.*
	Landsat data	U.S. Department of the Interior, Geological Survey, EROS Data Center (EDC), Sioux Falls, SD 57198, (605) 594-6151 (Aerial Photographs); and EOSAT, Landsat Customer Service c/o EROS Data Center (EDC), Sioux Falls, SD 57198, (605) 594-2291 (Landsat Data). *EDC sells high- and low-altitude photographs; and also reproduces and distributes Landsat data through a cooperative agreement with NOAA (National Oceanic and Atmospheric Administration) and EOSAT (Earth Observation Satellite Company).*
State		
California	Maps	NCIC State Affiliate: Map and Imagery Laboratory Library, University of California, Santa Barbara, CA 93106, (805) 961-2779.
Idaho	Maps	NCIC State Affiliate: Idaho State Historical Library, 610 N. Julia Davis Dr., Boise, ID 83702, (208) 334-3356.
Oregon	Maps	NCIC State Affiliate: Oregon State Library, Public Services, Salem, OR 97310, (503) 378-4368.
Washington	Maps	NCIC State Affiliate: Washington State Library, Information Services Division, Olympia, WA 98504, (360) 753-4027.

Table A-9. Sources for data on land-use history.

Data coverage	Data form	Source
Regional	Land-use maps	U.S. Department of the Interior, Geological Survey, National Cartographic Information Center (NCIC), Western Mapping Center, 345 Middlefield Road, Menlo Park, CA 94025, (415) 328-4309. *NCIC offices accept orders for aerial photographs and satellite imagery, and they sell custom cartographic products such as GS digital data, color and feature map separates, orthophotoquads, and land-use and land-cover maps.*
	County zoning mapslinfo	County planning and development departments. *These local departments may have zoning or land-use maps available. Zoning regulates acceptable uses for land and often is based generally on land-use history. These departments may also have current or historical aerial photographs, as well as geologic maps showing underlying geology and areas of unstable soils, GS topographic maps, and many other types of maps of that specific county. Contact your county's planning, zoning, or development department.*
	Maps/GIS digital	GS node of the National Geospatial Data Clearinghouse at http:llnsdi.usgs.govlnsdil. *Metadata that describe geology, water, and mapping sets are available on the World Wide Web. Many data sets are available on-line, including digital elevation model, land use/land cover, and water resources. The Land Use and Land Cover (LULC) data files describe the vegetation, water, natural surface, and cultural features on the land surface. The Geological Survey provides these data sets and associated maps as apart of its National Mapping Program.*
	National Resources Inventory	The National Resources Inventory (NRI) is an inventory, or catalog, of land-cover and use, soil erosion, prime farmland, wetlands, and other natural resource characteristics on nonfederal rural land in the United States. *See NRI annotations in Table **A-5,** Soils.* See also entries in Table A-7, Vegetation—historical.
State		
California	Maps	NCIC State Affiliate: Map and Imagery Laboratory Library, University of California, Santa Barbara, CA 93106, (805) 961-2779.
Idaho	Info	NCIC State Affiliate: Idaho State Historical Library, 610 N. Julia Davis Dr., Boise, ID **83702,(208)334-3356.**

Table A-9. Sources for data on land-use history.

Data coverage	Data form	Source
Oregon	Info	NCIC State Affiliate: Oregon State Library, Public Services, Salem, OR 97310, (503) 378-4368. The State of Oregon, through the Land Conservation and Development Commission (LCDC) set land-use zoning criteria for the entire State; each County developed a plan or code to comply with the LCDC guidelines. County Planning or Development Departments in Oregon can indicate how land is zoned. State of Oregon Land Conservation and Development Department, 1175 NE Court St., Salem, OR 97310, (503) 373-0050.
Washington	Info	NCIC State Affiliate: Washington State Library, Information Services Division, Olympia, WA 98504, (360) 753-4027.

Table A-10. Sources for precipitation data

Data coverage	Data form	Source
Regional	Maps/info	Oregon Climate Service, Strand Agricultural Hall, Room 316, Oregon State University, Corvallis OR, 97331-2209, (541) 737-5705, FAX (541) 737-2540, email to oregon@ats.orst.edu. George Taylor, State Climatologist. *Can provide hard-copy Annual Precipitation Map for the Western United States and States of Oregon, Washington, Idaho, Nevada, Montana, and Utah ($5.00 fee for 11" x 17" color-coded State map), and can provide other climatology information regarding a specific location. Phone responses and simple mailings have no fees. FAXs, diskettes, letters with data interpretation and GIS layers have fees. Data can be requested by phone, mail, or email. Access also provided to the NOAA atlas of precipitation frequency. World Wide Web homepage is on the Internet at http://ocs.ats.orst.edu, and from this page annual precipitation maps of Oregon and the Pacific Northwest can be downloaded. These maps were generated by the PRISM model by Chris Daly and are color coded in 100 mm increments.*
	Maps	New PRISM maps for the United States are now available via ftp. *These are 1961-1990 monthly mean precipitation grids, modeled at 2.5-min (~5 km) spatial resolution. Spatial domain is the lower 48 States. All but a few State maps are highly preliminary, but the peer-review and revision process is progressing. The maps will be updated periodically to reflect recent changes. To download the maps: (1) ftp fsl.orst.edu, (2) anonymous, (3) <your email address>, (4) cd pub/daly/prism, (5) binary, (6) get prism_us.Z (about 10 MB), (7) get prism_us.doc, (8) quit. Use the 'uncompress' command to extract the prism-us file. The file will expand to nearly 70 MB, so make sure you have disk space! Read the documentation carefully. It will indicate the status of the various State and regional sections of the maps, and provide important geographic information. The PRISM file is in a generic ASCII format that should be accessible by everyone.*
State		
Oregon	Maps	Oregon Climate Service, Strand Agricultural Hall, Room 316, Oregon State University, Corvallis OR, 97331-2209, (541) 737-5705, FAX (541) 737-2540, email to oregon@ats.orst.edu. (see above). *For Oregon, annual precipitation maps are available for each County.*
Washington	Maps	Washington Department of Natural Resources, Photo & Map Sales, P.O. Box 47013, Olympia, WA 98504-7013, (360) 902-1234. *Can provide Annual Precipitation Map for the State of Washington (fee).*

Table A-11. Sources for streamflow data.

Data coverage	Data form	Source
Regional		
USWEST	Digital	GS Gaging Station Records. Available on CD-ROM from USWEST Optical Publishing, Boulder, CO. Also available on the Internet via the World Wide Web at http://wwwdwatcm.wr.usgs.gov/historical.html for WA, OR, and ID.
	Info	GS National Water Data Storage and Retrieval System (WATSTORE). *All types of water data are accessed through WATSTORE, including an index of sites, daily water values with more than 240,000 daily parameters (e.g., streamflow, water temperatures, ground-water levels), peak flow file, water quality file, and ground-water site inventory file. Information on specific types of data, acquisition of data or products, and user charges can be obtained from the Wafer Resources Division District Offices (see State entries, below).*
	Maps/GIS digital	GS node of the National Geospatial Data Clearinghouse at http://nsdi.usgs.gov/nsdi. *Metadata that describe geology, water, and mapping sets are available on the World Wide Web. Many data sets are available on-line, including digital elevation model, land use/land cover, and water resources.*
State		
California	Info	GS Water Resources Division District Office, Federal Building, Room W-2235, 2800 Cottage Way, Sacramento, CA 95825, (916) 978-4633. *WATSTORE.*
	Info	California Department of Water Resources (DWR) World Wide Web site at http://wwwdtw.water.ca.gov. Division of Local Assistance (DLA) District Offices. Division of Local Assistance-Headquarters, Department of Water Resources, P.O. Box 942836, Sacramento, CA 94236-0001 or 1020-9th Street, Sacramento, CA 95814; Ray Hart, Division Chief (916) 327-1646. Northern District, 2440 Main Street, Red Bluff, CA 96080-2398; Linton A. Brown, District Chief (916) 529-7342. Central District, 3251 S Street, Sacramento, CA 95816-7017; Dennis Letl, District Chief (916) 445-5631. *Since 1988, when the Division was formed, staff has provided technical and financial assistance to State, Federal and particularly local agencies for developing, managing, and improving water resources in California. A variety of programs is offered through the headquarters office in Sacramento and the Northern, Central, San Joaquin, and Southern Districts located in Red Bluff, Sacramento, Fresno, and Glendale, respectively. The*

Table A-11. Sources for streamflow data.

Data coverage	Data form	Source
		Division has over 300 people working around the State who are skilled in various disciplines and who can answer questions on water quality, water rights, surface and ground water, geohydrology, desalination, reclamation and reuse of water, water conservation, land and water use, recreation planning, floodplain management, environmental review, agricultural drainage, water transfers, and long-range water supply and demand. The Division also administers loan and grant programs designed to restore urban streams and to make more efficient use of surface and ground water resources.
Idaho	Info	GS Water Resources Division District Office, 230 Collins Road, Boise, ID 83702, (208) 334-1750. *WATSTORE.*
	Info	Geological Survey (DOI) Idaho District homepage is the World Wide Web source for Idaho water information at http://wwwidaho.wr.usgs.gov. *Links to the Idaho District Water Data Page at http://wwwidaho.wr.usgs.gov/public/h2odata.html, which contains information on surface-water data, water-use data, the National Water Summary, national hydrologic conditions, and other resources. Includes an on-line Idaho District Data Requesf Form.*
	Info	The Idaho Department of Water Resources (IDWR) has its main office at 1301 North Orchard Road, Boise, ID, (208) 327-7900, FAX (208) 327-7866. IDWR has four full-service regional offices that can assist with water and adjudication matters. For help, contact *the* regional office in your area. Northern Region: 1910 Northwest Blvd., Suite 210, Coeur d'Alene, ID 83814-2615, (208) 769-1450, FAX (208) 769-1454; Western Region: 2735 Airport Way, Boise, ID 83705-5082, (208) 334-2190, FAX (208) 334-2348; Southern Region: 222 Shoshone St. East, Twin Falls, ID 83301-6105, (208) 736-3033, FAX (208) 736-3037; and Eastern Region: 900 North Skyline Drive, Idaho Falls, ID 83402-6105, (208) 525-7161, FAX (208) 525-7177. Also can be accessed via the World Wide Web at http://www.state.id.us/idwr/idwrhome.html. *The Idaho Department of Water Resources (IDWR) is responsible for the administration and allocation of water rights and permit and licensing systems to control beneficial use of Idaho waters. IDWR is also concerned with conservation and development of waters through planning, and it can provide information regarding endangered species, minimum streamflows, river flow information, floodplain management, stream channel alteration permits, etc.*
Oregon	Info	GS Water Resources Division District Office, 847 NE 19th Avenue, Suite 300, Portland, OR 97232, (503) 231-2009. *WATSTORE.*

Table A-11. Sources for streamflow data

Data coverage	Data form	Source
Washington	Info	GS Water Resources Division District Office, 1201 Pacific Avenue, Suite 600, Tacoma, WA 98402, (206) 593-6510. WATSTORE.
	Info	GS Water Resources Inventory. Public inquiries can be made of the GS regarding water resources of Washington State via the World Wide Web at http://wwwdwatcm.wr.usgs.gov/inquiries/html or via email (all Internet mail sent to pubinfo@maildwatcm.wr.usgs.gov will be delivered directly to the Public Information Officer). The on-line form allows messages to be sent to the Public Information Officer; questions and comments may also be sent to other GS contacts. Historical water resources data available and services provided by the Public Information Officer (PIO) include: • Loan copies of open-file reports, water-resources investigations, and water-supply bulletins for studies conducted in the State of Washington. • Limited loan copies are available on selected professional papers, water-supply papers, geohydrologic monographs, circulars, teachers' educational packets, techniques of water-resources investigations, miscellaneous field investigations, and hydrologic atlases. • Field measurement notes of streamflow for continuous, partial, and crest stage gage stations operated by the Tacoma Field Office. • Summary of field measurements of streamflow (mostly posf-1983 for Tacoma Field Office. For surface water unit values (transmission via satellite every 15, 30, or 60 minute values) of streamflow, gage height, reservoir elevation, or temperature preceding the current Water Year: all available data needs to be restored into the computer data base. A fee will be assessed based on the amount of restoration and review process needed. • Plots of streamflow peaks of interest. • Limited statistical analyses of flow duration, high- and low-flow frequency as well as peak flow frequencies. There is a fee for custom analyses. • Station description, quality of records, location of instrumentation, datum of gage, and remarks from old water supply papers. • Card catalogue information of old water quality data showing probable sources of unpublished data or data published in interpretive reports but not in dafa bases.

Table A-I1. Sources for streamflow data.

Data coverage	Data form	Source
		• Source programs of groundwater or surface models developed by the Washington District personnel. Information for all other GS modeling programs available to the public through Reston Headquarters or private companies. • Temperature records of selected gaging stations operated for State of Washington, Deparfment of Ecology. • Field data, notes, correspondence and other pertinent project records from Federal archives or the National Archives in response to on-going research studies, consulting services, and Freedom of Information Act requests. • Cross-sectional survey notes from streams and rivers where sediment studies have been done. • Forwards requests to GS Regional Office for certification of all types of records for official use in court testimonies. As NAWDEX (National Water Data Exchange) Assistance Center, responsibilities are limited to accessing GS data bases in the State of Washington. Requests for information contained in other State databases or related to other States are forwarded to the headquarters office of NAWDEX in Reston, Virginia. • Miscellaneous field measurements of streamflow in the State of Washington since 1890. There may be a charge for some requests, depending on the size and urgency.

Table A-12. Sources for data on stream and surface-water type.

Data coverage	Data form	Source
Regional		See entries under "Streamflow" above.
	Data	*FS Stream Survey Database. **Data are known to exist, but a specific source for the data is not known, the availability is not known, or the content of the data is not known.*
State		
California	Info	CA Department of Water Resources (DWR) World Wide Web site at http://wwwdrw.water.ca.gov. Division of Local Assistance (DLA) District offices. Division of Local Assistance-Headquarters, Department of Water Resources, P.O. Box 942836, Sacramento, CA 94236-0001 or 1020-9th Street, Sacramento, CA 95814; Ray Hart, Division Chief (916) 327-1646. Northern District, 2440 Main Street, Red Bluff, CA 96080-2398; Linton A. Brown, District Chief (916) 529-7342. Central District 3251 S Street, Sacramento, CA 95816-7017; Dennis Letl, District Chief (916) 445-5631. *See annotations above.*
Idaho	Info	Geological Survey (DOI) Idaho District homepage. World Wide Web Source for Idaho water information at http://wwwidaho.wr.usgs.gov. *Links to the Idaho District Water Data Page at http://wwwidaho.wr.usgs.gov/public/h2odata.html, which contains information on surface water data, water use data, the National Water Summary, national hydrologic conditions, and other resources. Includes an on-line Idaho District Data Request Form.*
	Info	Idaho Department of Water Resources (IDWR). IDWR has it's main office located at: 1301 North Orchard Road, Boise, ID, (208) 327-7900, FAX (208) 327-7866. IDWR has four full-service regional offices to assist with water and adjudication matters. For help, contact the regional office in your area. Northern Region: 1910 Northwest Blvd., Suite 210, Coeur d'Alene, ID 83814-2615, (208) 769-1450, FAX (208) 769-1454; Western Region: 2735 Airport Way, Boise, ID 83705-5082, (208) 334-2190, FAX (208) 334-2348; Southern Region: 222 Shoshone St. East, Twin Falls, ID 83301-6105, (208) 736-3033, FAX (208) 736-3037; and Eastern Region: 900 North Skyline Drive, Idaho Falls, ID 83402-6105, (208) 525-7161, FAX (208) 525-7177. Also can be accessed via the World Wide Web site at http:/lwww.state.id.uslidwr/idwr.home.html. *See annotations above.*

Table A-12. Sources for data on stream and surface-water type

Data coverage	Data form	Source
Oregon	Info	Oregon Department of Forestry, 2600 State Street, Salem, OR 97310, (503) 945-7200; Jim Brown, State Forester, (503) 945-7211; Mike Beyerle, Deputy State Forester, (503) 945-7202; Fred Robinson, Assistant State Forester, (503) 945-7205. • NORTHWEST OREGON AREA, Lee Oman, Area Director, Area Office, 801 Gales Creek Road, Forest Grove, Oregon 97116-1199, (503) 357-2191, FAX (503) 357-4548. District Headquarters: Forest Grove District, Dave Johnson, District Forester, 801 Gales Creek Road, Forest Grove, Oregon 97116-1199, (503) 357-2191, FAX (503) 357-4548); Tillamook District, Mark Labhart, District Forester, 4907 E. Third Street, Tillamook, Oregon 97141-2999, (503) 842-2545, FAX (503) 842-3143; Astoria District, Stan Medema, District Forester, Route 1, Box 950, Astoria, Oregon 97103, (503) 325-5451, FAX (503) 325-2756; Clackamas-Marion District, Dan Christensen, District Forester, 14995 S. Hwy. 211, Molalla, Oregon 97038, (503) 829-2216, FAX (503) 829-4736; West Oregon District, Mike Templeton, District Forester, 24533 Alsea Hwy., Philomath, Oregon 97370, (541) 929-3266, FAX (541) 929-5549; South Ford District, Fred Stallard, Administrative Supervisor, 48300 Wilson River Hwy., Tillamook, Oregon 97141, (503) 842-8439, FAX (503) 842-6572. • SOUTHERN OREGON AREA, Craig Royce, Area Director, Area Office, 1758 N.E. Airport Road, Roseburg, Oregon 97470-1499, (541) 440-3412, FAX (541) 440-3424. District Headquarters: Southwest Oregon District, Cliff Liedtke, District Forester, 5286 Table Rock Road, Central Point, Oregon 97502, (541) 664-3328, FAX (541) 776-6260; Coos District, Rick Rogers, District Forester, 300 Fifth Street, Bay Park, Coos Bay, Oregon 97420, (541) 267-4136, FAX (541) 269-2027; Western Lane District, Darrel Spiesschaert, District Forester, P.O. Box 157, Veneta, Oregon 97487-0157, (541) 935-2283, FAX (541) 935-0731; Eastern Lane District, Dan Shults, District Forester, 3150 Main Street, Springfield, Oregon 97478, (541) 726-3588, FAX (541) 726-2501; Linn District, Dan Shults, District Forester, 4690 Highway 20, Sweet Home, Oregon 97386, (541) 367-6108, FAX (541) 367-5613. • EASTERN OREGON AREA, Jeff Schwanke, Acting Area Director, Area Office, 3501 E. 3rd. Street, Prineville, Oregon 97754, (541) 447-5658, FAX (541) 447-1469. District Headquarters: Northeast Oregon District, Gary Rudisill, District Forester, 611 20th Street, La Grande, Oregon 97850, (541) 963-3168, FAX (541) 963-4832; Central Oregon District, Mike Howard, District Forester, 220710 Ochoco Hwy., Prineville, Oregon 97754, (541) 447-5658, FAX (541) 447-1469; Klamath-Lake District, Roy Woo, District Forester, 3400 Greensprings Drive, Klamath Falls, Oregon 97601, (541) 883-5681, FAX (541) 883-5555.

Table A-12. *Sources for data on stream and surface-water type.*

Data coverage	*Data form*	*Source*
		The Oregon Department of Forestry, authorized by Oregon Revised Statute 526.008 and established in 1911, is under the direction of the State Forester, who is appointed by the Oregon Board of Forestry. The statutes direct the State Forester to act on all matters pertaining to forestry in the protection of forest lands and the conservation of forest resources. The department administers the Oregon Forest Practices Act, Log Patrol Act, Log Brand Act, Small Tract Optional Tax Law, forest land classification, forestry assistance to Oregon's 24,000 non-industrial private woodland owners, forest resource planning, and community and urban forestry assistance. Staff can access data which identifies the type of surface water which may be present on a specific parcel of land. ** ODFW stream surveys of private land.* **Data is known to exist, but a specific source for the data is not known, the availability is not known, or the content of the data is not known.*
Washington	Map	*Washington Department of Natural Resources, Photo & Map Sales, P.O.* Box 47013, *Olympia,* WA 98504-7013, (360) 902-1234. *DNR Water Type Map.*
	Info	*Public inquiries of the Water Resources Inventory can be made to the GS about water resources of Washington State via the World Wide Web at* http://wwwdwatcm.wr.usgs.gov/inquiries/html *or via email (all Internet mail sent* to pubinfo@maildwatcm.wr.usgs.gov *will be delivered directly* to *the Public Information Officer).* *The on-line form allows messages to be sent to the Public Information Officer; questions and comments may also be sent to other GS contacts. See annotations above.*

Table A-13. Sources for water-quality data.

Data coverage	Data form	Source
Regional	Electronic	U.S. Department of Agriculture, National Resource Conservation Service, National Resources Inventory (NRI) at http://www.ncg.nrcs.usda.gov/nri.html.
State		
California		California Department of Water Resources (DWR), Division of Local Assistance (DLA) District Offices. Division of Local Assistance-Headquarters, Department of Water Resources, P.O. Box 942836, Sacramento, CA 94236-0001; or 1020-9th Street, Sacramento, CA 95814; Ray Hart, Division Chief (916) 327-1646. Northern District, 2440 Main Street, Red Bluff, CA 96080-2398; Linton A. Brown, District Chief (916) 529-7342. Central District 3251 S Street, Sacramento, CA 95816-7017; Dennis Letl, District Chief (916) 445-5631.
		U.S. Department of the Interior, Geological Survey, Water Resources Division District Office, Federal Building, Room W-2235, 2800 Cottage Way, Sacramento, CA 95825, (916) 978-4633.
		California Department of Water Resources (DWR) at http://wwwdrw.water.ca.gov.
Idaho		Idaho Department of Water Resources (IDWR). IDWR main office: 1301 North Orchard Road, Boise, ID, (208) 327-7900, FAX (208) 327-7866. IDWR has four regional offices: Northern Region: 1910 Northwest Blvd., Suite 210, Coeur d'Alene, ID 83814-2615, (208) 769-1450, FAX (208) 769-1454; Western Region: 2735 Airport Way, Boise, ID 83705-5082, (208) 334-2190, FAX (208) 334-2348; Southern Region: 222 Shoshone St. East, Twin Falls, ID 83301-6105, (208) 736-3033, FAX (208) 736-3037; and Eastern Region: 900 North Skyline Drive, Idaho Falls, ID 83402-6105, (208) 525-7161, FAX (208) 525-7177.
		U.S. Department of the Interior, Geological Survey, Water Resources Division District Office, 230 Collins Road, Boise, ID 83702, (208) 334-1750.
		Idaho Department of Water Resources at http://www.state.id.us/idwr/idwrhome.html.
Oregon		U.S. Department of the Interior, Geological Survey, Water Resources Division District Office, 847 NE 19th Avenue, Suite 300, Portland, OR 97232, (503) 231-2009.
		Oregon Rivers Information System (ORIS), Northwest Environmental Database, Brent O. Forsberg, Coordinator at forsberg@dfw.or.gov. ORIS can be accessed through the Oregon Department of Fish and Wildlife (ODFW), homepage at http://www.dfw.state.or.us.
Washington		Washington Department of Natural Resources, Photo & Map Sales, P.O. Box 47013, Olympia, WA 98504-7013, (360) 902-1234.U.S. Department of the Interior, Geological Survey, Water Resources Division District Office, 1201 Pacific Avenue, Suite 600, Tacoma, WA 98402, (206) 593-6510.
		U.S. Department of the Interior, Geological Survey, Water Resources Division District Office, 1201 Pacific Avenue, Suite 600, Tacoma, WA 98402, (206) 593-6510.
		———Geological Survey, Water Resources Inventory; Water Resources of Washington State at http://wwwdwatcm.wr.usgs.gov/inquiries/html.

Table A-14. Sources for data on fish species distributions.

Data coverage	Data form	Source
Regional	GIS	*EPA *River Reach Database.* **Data is known to exist, but a specific source for the data is not known, the availability is not known, or fhe content of the data is not known.*
	Maps	U.S. *Department of the Interior, Geological Survey (GS).* *GS markets four series of maps depicting the distribution of certain fish and wildlife species and other ecological elements along the coastal areas of the conterminous 48 States. Produced by the Fish and Wildlife Service (FWS) from GS base data, the maps are designed fo help in making location and design decisions about development along the coasts. The maps cover broad geographic areas with limited topographic detail and depict the habitats of fish and wildlife. Of particular interest are the coastal habitats of endangered species, migratory waterfowl, and commercially important fish. The maps also show certain land-use designations, such as national wildlife refuges, State waterfowl management areas, and parks. The five-color maps are printed on 24- by 35-inch sheets, each covering 2 degrees of longitude by 1 degree of latitude. The Pacific Coast maps are the first comprehensive series of natural resource maps of the West Coast. The maps depict fish and wildlife and their habitats and major land-use designations. The 30-map series covers the entire 40,150 square-mile Pacific coastal zone from Mexico to Canada, including Puget Sound. The 159-page narrative report provides detailed explanations and additional technical information about the ecological data displayed on each map. The ecological data plotted on the maps is derived from FWS ecological inventories. These maps can be obtained from any of the GS map sources listed under "Topography," above.*
State		
California	*GIS*	*California Department* of *Fish and Game.* For *further information about the Geographic Information System contact* John *Ellison,* 1730 I *Street, Suite* 100 *Sacramento, CA* 95814, (916) 323-1477, *email* to *jellison@dfg.ca.gov.*
Idaho	*Info*	*Idaho Department* of Fish *and Game. Headquarters,* 600 *S. Walnut, P.O. Box 25, Boise, ID* 83707, (208) 334-3700; *Panhandle Region,* 2750 *Kathleen Avenue, Coeur* d'Alene, ID 83814, (208) 769-1414; *Clearwater Region,* 1540 *Warner Avenue, Lewiston,* ID 83501, (208) 799-5010; *Southwest Region,* 3101 *S. Powerline Road, Nampa, ID* 83686, (208) 465-8465 *(From Boise, call:* 887-6729); *McCall, 555 Deinhard Lane, McCall,* ID 83638, (208) 634-8137; *Magic Valley Region,* 868 *East Main Street,* P.O. *Box* 428, *Jerome, ID* 83338, (208) 324-4350; *Southeast Region,* 1345 *Barton Road, Pocatello, ID* 83204, (208) 232-4703; *Upper Snake Region,* 1515 *Lincoln Road, Idaho Falls, ID* 83401, (208) 525-7290, Salmon Region, 1214 **Hwy** *93 N.,* P.O. Box 1336, *Salmon, ID* 83467, (208) 756-2271. Also *available is a homepage on the World Wide Web at* http://www.state.id.uslfishgame/fkhgame.html.

Table A-14. *Sources for data on fish species distributions.*

Data coverage	*Data form*	*Source*
Oregon	*Data*	*Oregon Rivers Information System (ORIS)—Northwest Environmental Database. Coordinator: Brent O. Forsberg; email to forsberg@dfw.or.gov.* ORIS *can be accessed on-line on the World Wide Web through the ODFW homepage at* http://www.dfw.state.or.us). *The Oregon Rivers Information System is a comprehensive collection of data on the rivers in the State of Oregon. The data is part of a four-State collection effort by the Bonneville Power Administration called the Northwest Environmental Database. These other States include Washington, Idaho, and Montana. The search program allows the user to view data on the following Oregon river resources: anadromous fish, resident fish, wildlife, natural features, recreation, cultural features, institutional constraints, and other associated resources. The user will be presented with a series of menus allowing searches by a specific river, a drainage basin, or a county of interest; a specific resource type in any drainage basin or county; a specific township and range for resources; and a specific river reach by Environmental Protection Agency reach number. By selecting one of the on-line options, you may read the Operation Manual; down load the search program and data files; or use the ORIS program to search data on-line.*
	Data	**Oregon Department* of *Fish and Wildlife (ODFW), Oregon Species Information Database. ODFW, Northwest Region, Corvallis, OR,* (541) 757-4186. *Contact: Wanda* McKenzie. **Data is known to exist, but a specific source for the data is not known, the availability is not known, or the content of the data is not known.*
	Data	*Oregon State University Museum of Ichthyology. Corvallis, OR. Contact Dr. Doug Markle,* (541) 737-1970.

Table A-15. Sources for data on threatened and endangered species (fish and other biota).

Data coverage	Data form	Source
Regional	Lists	U.S. Department of the Interior, Fish and Wildlife Service, Portland, OR (503) 231-6118. FWS also maintains a homepage on the World Wide Web with a sub-directory for lists of endangered species in Region 1 (includes Pacific Northwest) at http://www.fws.gov/statl.rl.html. *Lists are maintained by the FWS; they include endangered and threatened species under FWS jurisdiction, and species listed (added) under a memorandum of understanding between FWS the U.S. Department of Commerce, National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (NMFS).* *Species or stocks under the sole jurisdiction of NMFS are not listed. For freshwater habitats in the Pacific Northwest, those species/stocks are Sacramento R. winter run chinook salmon; Snake R. spring/summer run chinook salmon; Snake R. fall run chinook salmon; Snake R., (ID, stock) sockeye salmon: Shortnose sturgeon* (Acipenser brevirostrum).
State		
California	Lists	California Department of Fish and Game, Natural Diversity Data Base (NDDB). The NDDB can be accessed through the World Wide Web at http://spock.dfg.ca.gov/Endangered/endangered.html, and it provides lists of endangered and threatened species. *The NDDB is constantly being updated and expanded. All locational data entered into the NDDB are based on actual field sightings. There is an on-line disclaimer "The absence of a listed species from the county accounts does not necessarily mean it is absent from the county, only that no occurrence data are currently entered into the NDDB. Data from the Data Base does not constitute an official response from a State agency, will not in itself meet the requirements of the California Environmental Quality Act and does not replace the need for conducting field work." There are several categories of endangered species. Both Federal and State categories are defined and listed.*
Idaho	Info	Idaho Department of Fish and Game. Headquarters, 600 S. Walnut, P.O. Box 25, Boise, ID 83707, (208) 334-3700; Panhandle Region, 2750 Kathleen Avenue, Coeur d'Alene, ID 83814, (208) 769-1414; Clearwater Region, 1540 Warner Avenue, Lewiston, ID 83501, (208) 799-5010; Southwest Region, 3101 S. Powerline Road, Nampa, ID 83686, (208) 465-8465 (from Boise call 887-6729); McCall, 555 Deinhard Lane, McCall, ID 83638, (208) 634-8137; Magic Valley Region, 868 East Main Street, P.O. Box 428, Jerome, ID 83338, (208) 324-4350; Southeast Region, 1345 Barton Road, Pocatello, ID 83204, (208) 232-4703; Upper Snake Region, 1515 Lincoln Road, Idaho Falls, ID 83401, (208) 525-7290; Salmon Region, 1214 Hwy 93 N., P.O. Box 1336, Salmon, ID 83467, (208) 756-2271. *Also* has a homepage *on* the World Wide Web at http://www.state.id.us/fishgame/fishgame.html.

Table A-15. Sources for data on threatened and endangered species (fish and other biota).

Data coverage	Data form	Source
	Info	Idaho Department of Water Resources. Main office: 1301 North Orchard Road, Boise, ID, (208) 327-7900, FAX (208) 327-7866. IDWR has four full-service regional offices to assist with all of water and adjudication matters. For, contact the regional office in your area. Northern Region: 1910 Northwest Blvd., Suite 210, Coeur d'Alene, ID 83814-2615, (208) 769-1450, FAX (208) 769-1454; Western Region: 2735 Airport Way, Boise, ID 83705-5082, (208) 334-2190, FAX (208) 334-2348; Southern Region: 222 Shoshone St. East, Twin Falls, ID 83301-6105, (208) 736-3033, FAX (208) 736-3037; and Eastern Region: 900 North Skyline Drive, Idaho Falls, ID 83402-6105, (208) 525-7161, FAX (208) 525-7177. Also, IDWR can be accessed via the World Wide Web at http://www.state.id.us/idwr/idwrhome.html. *The Idaho Department of Water Resources (IDWR) is responsible for the administration and allocation of water rights and permit and licensing systems to control beneficial use of Idaho waters. IDWR is also concerned with conservation and development of waters through planning, and can provide information regarding endangered species, minimum streamflows, river flow information, floodplain management, stream channel alteration permits, etc.*
Oregon	Info	Oregon Department of Fish and Wildlife (ODFW), 2501 SW First Ave., P.O. Box 59; Portland, OR 97207. General Phone Number: (503) 229-5406; General Information: (503) 229-5222; Habitat Conservation Division: (503) 229-6967; Wildlife Division: (503) 229-5454. ODFW Regional Offices: Northwest Region, Corvallis, (541) 757-4186; Southwest Region, Roseburg, (541) 440-3353; Central Region, Bend, (503) 388-6363; Northeast Region, LaGrande, (541) 963-2138; Southeast Region, Ontario, (541) 573-6582; Marine Region, Newport (541) 867-4741; Columbia Region, Clackamas, (503) 657-2000. ODFW can be accessed through the World Wide Web at http://www.dfw.state.or.us. The Executive Summary of the *Biennial Report on the Status of Wild Fish in Oregon* is available and can be downloaded from this source. *This summary provides an overview of selected anadromous and game fish species of concern and their locations, as well as a table of Oregon endangered, threatened, and sensitive nongame fishes. The executive summary addressed the status of selected species while the full report includes information on all wild freshwater and estuarine fish species in Oregon. Most of the information in the report comes from ODFW files, particularly annual reports filed by ODFW district biologists or from State research projects For more information about this report contact Kathryn Kostow at ODFW, email to kostowk@dfw.or.gov.*

A.3 Laws and Regulations

By means of laws and regulations, cooperative leadership and funding are provided to States and local landowners to acconiplish stated goals of the laws and their programs. These laws, regulations, and programs, in turn, need scientific information to accomplish their objectives. Moreover, many of the laws incorporate clauses that emphasize the importance of citizens and recognize that environments diverse and safe for other living things provide healthy physical and economic environments for people.

Four sets of rules need to be examined to understand the effect of laws and regulations on salmonid habitat: tlie U.S. Code (which include the public laws and the statutes), Federal regulations (pursuant to the USC, laws, and statutes), state codes (statues), and state regulations.

Federal legislation is developed when Congress passes bills; after signing by the President (or tlie override of a veto), the bill becomes law. First published in "slip" form (usually saddle-stapled sheets), it is called a Public Law and is given a number that designates the session of Congress and then the sequential order in which the bill was signed into law. *Statutes At Large* are bound collections of Public Laws ordered sequentially; the *U.S. Code* (USC) integrates laws and their amendments with related laws by subject into bound volumes (called titles) that are periodically updated. Unfortunately, the sections of each particular law are numbered differently from one form to the next; also, all public laws do not ultimately become published in tlie USC. Laws transfer power ("authorize") and they designate levels of funding ("appropriate"); often one law authorizes certain action and suggests a level of support while another law actually appropriates funds—the root cause of so-called "unfunded mandates." Laws may be adjudicated in civil or criminal courts.

Federal regulations originate in the Executive branch as a response by the department or agency authorized to implement a particular public law; these regulations are usually published first in the *Federal Register* for public comment. Regulations are revised, republished as final, and ultimately codified—collected, bound, and published—in the *Code of Federal Regulations* (CFR). Regulations make laws operative, they have the force of law, and their purposes are administrative or related to enforcement. In addition to civil and criminal courts, regulations may also be adjudicated in the Federal administrative court system.

Forms and function of State laws and regulations follow those of the Federal government: State laws originate in legislatures and are collected into books of statutes; regulations are then promulgated by State executive agencies to iniplement the laws. Access to the law largely depends on understanding its structure, purpose, and function.

In many ways, laws and regulations prescribe ideals. Courts, however, play a pivotal role in applying law to actual situations: court cases and decisions shape the interpretation and direct the meaning a law assumes over time. Regardless of legislative intent and executive management goals, law can come to mean what the courts say it means, a result of selected information, situational evidence, savvy argument, and persuasion. These meanings are socially derived, and they have come to be one documented expression of social values tied to a particular time and place. Unlike the law, court cases clarify how people will act (not how they should act) with respect to property, land, other people, other species, and so on. As a result, an accurate analysis of how law and regulations effect salmon habitat niust ultimately review decisions of the civil, criminal, and administrative courts of both State and Federal systems.

This section lists and describes briefly Federal and State laws related to salmon habitat restoration both implicitly and explicitly. References to regulations have been collected only when they were encountered; further research and analysis in this literature would yield prescribed practices. Finally, court case literature would yield information on whether practices in laws and regulations were accepted and applied or were challenged and changed. We do not review case law herein.

A.3.1 Federal Laws

The pre-eminent Federal laws used to protect salmonids and their habitats include the Endangered Species Act (ESA), tlie Clean Water Act (CWA), the National Environmental Policy Act (NEPA), and the Food Securities Act (FSA). ESA was created to conserve the ecosystems upon which threatened and endangered species depend and to provide a program to conserve listed species and their ecosystems. Various sections of ESA obligate Federal agencies to minimize putting listed species in further jeopardy, and it outlines permit conditions including take. CWA is intended to restore and maintain the chemical, physical, and biological integrity of the Nation's waters by eliminating the discharge of pollutants into waters and by attaining water quality suitable for fish and wildlife. EPA has developed guidelines that decree protection from discharges from agriculture, forestry, mining, construction and hydrologic modifications. NEPA has a policy section that identifies the rights and responsibilities of each person to enjoy, preserve, and enhance the environment. The Federal government is responsible for coordinating Federal programs to help people

preserve a diverse environment aiid act as trustees for future generations. FSA provides incentives for farmers and ranchers to conserve riparian areas and wetlands in order to continue receiving Federal subsidies. Each of these four laws is discussed in greater detail in Part I, Chapter 9 of this document.

Other Federal laws explicitly extended to private landowners include the Forest Stewardship Act (PL 102-574, 16 USC 1600 et seq.), which amends the Federal Forest and Rangeland Renewable Resources Planning Act (PL 102-574, 16 USC 2101), and the Coastal Zone Management Program (CZMP). The Forest Stewardship Act allows local foresters to develop a program for management of nonfederal lands, and the CZMP was developed to protect beneficial public uses including biological resources and water quality, but it does not apply to streams with flow less than 20 cfs. Three other relevant Federal programs are tlie Conservation and Wetland Reserve Programs, which compensate farmers who protect sensitive lands by removing them from production, and the Surface Mining Control aiid Reclamation Act (PL 98-409), which regulates open pit mining.

A.3.2 State Laws

Numerous State laws and programs have been enacted that directly or indirectly relate to the protection of salmonids (and other fishes) and their habitats or other beneficial uses of streams and rivers. The following section briefly describes laws and regulations related to general land use, forestry, agriculture and pesticides, range, mining, water quality, instream flows, and channel alteration.

California

The sources for most of our legal information for California was West's Annotated California Codes (WACC).

General Land Use. One of the most far reaching laws is the California Coastal Act (CCA, Public Resources Sections 30,000-31,405), which creates state-local partnerships for comprehensive land-use planning. The CCA requires protection of public access to the shore, conservation of environmentally sensitive habitats, and preservation of scenic beauty through development restrictions.

Forestry. Forest Practices in California are mandated by tlie Z-Berg–Nejedly Forest Practices Act (1973). California's Forest Practice Rules (Title 14, Subchapters 4–6, California Code of Regulations) covers silvicultural methods (Article 3), harvest practices and erosion control (Article 4), site preparation (Article 5), water course and lake protection (including riparian protection zones; Article 6), and roads (including water crossings; Article 12).

Agriculture. Pesticide uses are restricted by Food and Agriculture Sections 12971-12979, while pesticide monitoring is covered in Section 13148. Screens at diversions are also required (Fish & Game 5900–6028).

Mining. Permits are required for suction dredging (Fish and Game Section 5653), placer mining requires pollution controls (Public Resources Section 2555), and protection and reclamation of mined land is ensured (Public Resources Section 2710).

Endangered Species. State endangered and threatened species are protected (Public Resources Code Section 2050) and public funds (separate from fish and game or nongame funds) are authorized for native species conservation and enhancement (Fish and Game Section 1750).

Water Quality. Water quality laws are outlined in two areas. Fish and Game Section 5650 makes it "unlawful to deposit in, permit to pass into, or place where it can enter waters, any material deleterious to fish, plant, or bird life." The Water Quality Control Act (Water Section 13,000 et seq.) authorizes standards for point and diffuse pollution, combines quality and quantity issues, requires permits for dischargers, including dredging and filling (Section 13,376). Unpermitted discharges are subject to civil penalties (Section 13,385), while intentional or negligent violations are subject to criminal penalties (Section 13,387). Section 13,050 defines a waste as any waste substance associated with human habitation or of human or animal origin. Pollution includes wastes that unreasonably affect beneficial uses, while beneficial uses include recreation, esthetic enjoyment, and preservation or enhancement of fish, wildlife, or other aquatic resources. This Act is available on the internet:
http://agency.resource.ca.gov/wetlands/permitting/tbl_cntnts_porter.html.

Instream Flows. California Fish and Game sets minimum flows to assure continued viability of stream fish and wildlife (Public Resources Code Section 10001).

Channel Alterations. Devices that prevent or impede fish passage, or tend to do so, are prohibited (Fish and Game Sections 5901 and 12015). Additional protections against channelization and other disturbances of the bed, bank, and channel are covered in Public Resources Code Section 1600 et seq.

Idaho

Relevant laws for Idaho were gleaned from tlie Idaho Code.

Forest Practices. Rules and regulations pertaining to the Idaho Forest Practices Act, Title 38, Chapter 13, Idaho Code are given in Idaho Administrative Rules IDAPA 20.15 -- Department of State Lands. Rule 2 includes general rules. Rule 3 regulates timber harvest activities, including those in riparian areas. Rule 4 prescribes restrictions for stream crossings. If stream beneficial uses are not fully protected aiid the activity is deemed a substantial threat, the activity can be halted (Section 38-1314).

Agriculture. Pesticide restrictions are outlined in Section 22-3420. Fish screens are required on irrigation diversions (Section 36-906).

Mining. Surface mining is regulated under tlie Idaho Surface Mining Act (Title 47, Chapter 15, Idaho Code). Tlie purpose of the Act is to protect tlie public health, safety, and welfare by requiring reclamation of all lands disturbed by mineral exploration and surface mining operations (Section 47-1501). It requires the operator to, among other things, provide maps and diagrams of tlie mining site identifying access and haul roads, nearby creeks or other water bodies, mining pits, mineral stockpiles, and tailings, as well as to develop a reclamation plan (Section 47-1506). Dredge and placer mining must also be conducted in a manner that protects stream and watercourses for the enjoyment , use, and benefit of all people (Section 47-1312)

Water Pollution. Existing instream beneficial uses of each water body and the level of water quality necessary to protect those uses must be maintained aiid protected (Section 39-3601 to 3603). Water pollution is defined as alteration of tlie physical, thermal, chemical, or biological properties of State waters that will (or is likely to) render waters detrimental to recreational and esthetic uses or to fish or aquatic life (Section 39-103). Tlie State has tlie authority to enter private property to conduct monitoring.

Instream flows. Minimum flow is considered a beneficial use to protect fish, wildlife habitat, aquatic life, water quality, esthetics, or recreation. Minimum flows are the amounts of water needed to protect such uses (Section 42-1501).

Channel Alterations. No person may construct or maintain a dam or other obstruction without installing a proper fishway (Section 36-906). Fish screens are required on all canals aiid conduits (Section 36-906). Unpermitted channel alterations are prohibited (Section 42-3801-3813) and they are also restricted by the general nuisance law (Section 52-101-111).

Oregon

Laws and regulations for Oregon were taken from the Oregon Revised Statutes (ORS), the Oregon Administrative Rules (OAR), and Butterwortli's *Oregon Revised Statutes Annotated.*

General Land Use. All land in Oregon is zoned by counties to meet land use criteria set by the Land Conservation and Development Commission. Zoning is designed to protect forest and agricultural land from residential developments and tlie Oregon coast from recreational home developments that preclude easy public access to the State's beaches and estuaries. OAR 603-70 and 72 grant funds to surface water, flood, and municipal districts for erosion control, water conservation, water quality enhancement, stream bank stabilization, and riparian management projects. Substantial damage to wildlife, flora, aquatic or marine life, or habitat is considered an environmental crime (ORS 468.920).

Forestry. Oregon's Forest Practices Act (44 ORS 527-610–770) mandates protection of fishery resources during forestry activities. Rules for channel alterations, riparian conditions, chemical application, harvest, road construction and maintenance, and forested wetlands are described in OAR 629. OAR 629-57-2000–2660 specifically address water protection. The Board of Forestry directed that monitoring of water quality and fish habitat receive high priority and adequate funding (OAR 629-57-2005). Forest practices rules require that stream crossing structures provide passage for adult and juvenile fish, both up- and downstream (OAR 629-24-522).

Agriculture. Pesticide restrictions are outlined in OAR 603-57. Basins designated as total maximum daily load (TMDL) waters are subject to water quality management plans (ORS 568.900-.933). These plans are designed to prevent and control water pollution from agriculture by restricting land clearing and cropping practices. Confined animal operations are regulated under OAR 340-51, which prohibits animal wastes from waters without a permit and requires manure to be collected, stored, and distributed so as to prevent pollution. Civil penalties are assessed for failure to submit plans, violation of permit compliance schedules, failure to provide access, placing wastes where they are likely to enter waters, unpermitted discharges, water pollution, standards violations, or use impairment. OAR 603-90 protects water uses required by State and Federal law. Adversely affected water uses are listed, the necessary pollution control measures are described, and a strategy and schedule for iniplenientation are developed. Violations are the same as for confined

animal operations and each day's violation is considered a separate offense. Fish screens are covered in 41 ORS 498.705–750.

Rangelands. This regulation is designed to restore properly functioning ecosystems and ecosystem processes on State lands by maintaining, restoring, or enhancing water quality and rangeland health (OAR 141-110). Health is defined as soil integrity and sustainable ecological processes. Rangelands vulnerable to adverse transitions are to be monitored.

Mining. A permit is required for chemical process mining (OAR 690-78). It must depict the duration, location, diversions, and measures to avoid damaging aquatic life or public water uses.

Water Quality. ORS 468B defines pollution as the alteration of physical, chemical, or biological properties of waters, including temperature and turbidity that renders such waters detrimental to fish, aquatic life, or their habitat. It also prohibits pollution, placing wastes where they are likely to be carried to waters, and discharge of wastes if they reduce water quality standards. Water quality violations include causing major harm or risk to the environment and failure to provide access when required (OAR 340-12-055). Detrimental change in biological communities is prohibited (OAR 340-41-027). Miscellaneous provisions on water rights, uses, and protection are treated in 45 ORS 541.605.990, chapter 541.

Instream Flows. OAR 340-56 allows the Department of Environmental Quality (DEQ) to apply for instream water rights for pollution abatement, while OAR 690-77 allows DEQ, the Department of Fish and Wildlife, and the Parks Department to obtain instream rights for aquatic life, recreation, wildlife, ecological values, and pollution abatement. Water control structures and water diversions are not required to obtain these rights.

Channel Alterations. Removal and fill for all waters and wetlands of the State requires a permit and plan (OAR 141-85). The plan must describe the public value of the project, its duration, hydrological and fish impacts, and potential effects on rare, threatened or endangered species. Wetland fill and removal requires mitigation that exceeds or equals the value of the wetland.

Washington

Laws for this State are summarized primarily from *West's Revised Code of Washington, Annotated.*

General Land Use. The State Growth Management Act (WAC 365-195) requires counties with human populations of 50,000 or more to develop plans for urban growth following standards. Urban growth areas can also be designated in rural areas. The Shoreline Management Act (90.58 RCW) protects State over local interests, long- versus short-term issues, shoreline ecology, increased public access, and recreational values through land-use planning.

Forestry. Timber harvesting regulations, including riparian protections, are detailed in WAC 222-30. WAC 222-24 describes requirements for road construction and maintenance, including stream crossings (WAC-24-040). Use of forest chemicals is covered in WAC 222-38. Sections 76.42.030 RCW and 76.42.060 RCW, respectively, authorize wood debris removal and prohibit its deposition in channels. **A** program to reduce hazards from mass earth movements by identifying sensitive sites and restricting uses is described in 76.09.300–320 RCW. Tlie Department of Ecology can modify forest practice regulations (90.48.420 RCW) that result in pollution. The Departments of Forestry and Ecology have right of entry at any reasonable time (76.09.150, 160 RCW).

Agriculture. Pesticide uses and users are restricted (15.58 RCW). Tlie Department of Ecology can issue pollution violation notices for agricultural activities (90.48.450 RCW).

Mining. Section 78.56 RCW requires an environmental impact statement before mining. The Departments of Ecology and Fish and Wildlife incorporate mitigation measures in the permit to reduce impacts on fish and wildlife. Mine and mill tailings and effluents must be reduced by stabilization, removal, or reuse. Quarterly inspections are required and citizen reviews and suits are allowed. Aggregate mining is covered under WAC-220-110-130.

Water Quality. The State Water Protection Act (WAC 173-20) requires that beneficial uses of water be maintained and allows no further degradation of these uses. Pollution is defined as alteration of the physical, chemical, or biological environment, including temperature, turbidity or any substance likely to be detrimental to fish and aquatic life (90.48.020 RCW). Pollution is unlawful (90.48.080 RCW), including that from fish hatcheries (90.48.210 RCW), and entry rights are provided (90.48.090 RCW). **A** coastal protection fund is authorized (90.48.390-400 RCW).

Instream Flows. The Department of Ecology may establish minimum flows to protect fish, esthetics, recreation, and water quality (90.22.010 RCW). Water flows are also covered in 75.20.50 RCW.

Channel Alterations. The Hydraulics Code provides guidelines for bank protection (WAC 220-110-050), dredging (WAC 220-110-130), treatment of large woody material (WAC 220-110-150), and

culvert installation (WAC-220-1 10-070). Channel obstructions are prohibited on waters that are boatable or that can float logs or posts (88.28.050 RCW) and fishways are required around dams or obstructions (72.20.060 RCW). Section 86.16 RCW regulates construction and operations in floodplains that adversely influence flow regimes or health and property. Practices on aquatic lands must preserve and enhance water dependent uses, giving nonwater dependent uses low priority (79.90.450-545 RCW). Wildlife habitat and spawning values must be considered before leasing.

A.4 Federal and State Government Offices

Below are addresses, phone numbers, FAX numbers, and internet addresses for Federal and State Agencies that may be able to provide assistance or data related to conservation planning. Because electronic information changes frequently, access to the World Wide Web (URLs), email addresses, and telephone area codes with nunibers below inay have changed.

A.4.1 Federal Offices

U.S. Departmenf of Agriculture.

———Forest Service, 316 E. Myrtle, Boise, ID 83702, (208) 364-4340.

———Forest Service, 630 Sansome St., San Francisco, CA 94111 (415) 556-8551.

———Forest Service, 319 SW Pine, Portland, OR 97208 (503) 221-3418.

———Natural Resources Conservation Service, National Cartography and Geospatial Center, 501 Felix St., Bldg. 23 (P.O. Mail 6567), Fort Worth, TX 761 15, (800) 672-5559.

———Natural Resources Conservation Service, National Soil Survey Center, Soil Survey Laboratory, Federal Building, Room 152, MS 41. 100 Centennial Mall North, Lincoln, NE 68508-3866, (402) 437-5363.

———Natural Resources Conservation Service, 101 SW Main Suite 1300, Portland, OR 97204 (503) 414-3094.

———National Resource Conservation Service (California), 2121-C Second Street, Suite 102, Davis, CA 95616 (916) 757-8262

———Natural Resources Conservation Service, Resources Inventory and Geographic Information Systems Division, P.O. Box 2890. Washington, D.C. 20013, (202) 720-4530.

U.S. Departmenf of Commerce.

———National Marine Fisheries Service, 3773 Martin Way E., Building C, Olympia, WA 98501, (360) 534-9330.

———National Marine Fisheries Service, 525 NE Oregon St., Portland, OR 97232-2737, (503) 230-5400.

———National Marine Fisheries Service NW Regional Office, 7600 Sand Point Way, N.E.BIN C15700, Bldg. 1, Seattle, WA 981 15-0070, (206) 526-6150.

———National Marine Fisheries Service, Boise Field Office, 1387 S. Vinnel Way, Ste 377, Boise, Idaho, 83709, (208) 378-5696.

———National Marine Fisheries Service, SW Region, 501 West Ocean Blvd., Ste 4200, Long Beach, CA 90802-42 13, (310) 980-4001

———National Marine Fisheries Service, Santa Rosa Field Office, 777 Sonoma Ave., Rm 325, Santa Rosa, CA 95404-6515, (707) 575-6050

U.S. Department of the Inferior.

———Bureau of Land Management, 825 N. E. Multnomah, Portland, OR 97208, (503) 952-6002 (OR & WA).

———Bureau of Land Management, 316 E. Myrtle, Boise, ID 83702, (208) 364-4340.

———Bureau of Land Management, Federal Office Building Room E-2841, 2800 Cottage Way, Sacramento, CA 95825, (916) 484-4676.

———Fish and Wildlife Service, 500 NE Multnomah Suite 1692, Portland, OR 97232, (503) 231-6118 (CA, ID, OR, WA).

———Fish and Wildlife Service, Klamath River Office (California), 1215 S Main, Suite 212, Yreka, CA 96097-1006, (916) 842-5763.

———Fish and Wildlife Service, Coastal Office (California), 1125 16th St., Room 209, Arcata, CA 95521-7201, (707) 822-7201.

———Fish and Wildlife Service, Idaho State Office, 4696 Overland Rd., Room 576, Boise, ID 83705, (208) 334-1931.

———Fish and Wildlife Service, Oregon State Office, 2600 SE 98th Ave., Suite 100, Portland, OR 97266, (503) 231-6179.

———Fish and Wildlife Service, HCP-Forest Resources (Oregon), 333 SW 1st Ave., Portland, OR 97208-3623, (503) 326-62 18.

____Fish and Wildlife Service, Consultation & Conservation Planning (Oregon), 911 NE 11th Ave. Portland, OR 97232-4181, (503) 231-6241..

____Fish and Wildlife Service, HCP Program, 3704 Griffin Lane, Suite 102, Olympia, WA 98501, (360) 753-4474.

____Fish and Wildlife Service, Upper Columbia River, 11103 E. Montgomery Dr., Suite 2, Spokane, WA 99206, (509) 891-6839.

____Geological Survey, EROS Data Center (EDC), Sioux Falls, SD 57198, (605) 594-6151 (Aerial Photographs); and EOSAT, Landsat Customer Service c/o EROS Data Center (EDC), Sioux Falls, SD 57198, (605) 594-2291 (Landsat Data).

____Geological Survey, Geological Inquiries Group (GIG), 907 National Center, Reston, VA 22092, (703) 648-4383.

____Geological Survey, Map Distribution Section (MDS), Map Sales, Federal Center, Box 25286, Denver, CO 80225, (303) 236-7477.

____Geological Survey, National Cartographic Information Center (NCIC),Western Mapping Center, 345 Middlefield Road, Menlo Park, CA 94025, (415) 328-4309.

____Geological Survey, Photographic Library (LIB-P), Mail Stop 914, Building 20, Box 25046, Federal Center, Denver. CO 80225, (303) 236-1010.

____Geological Survey, Public Inquiries Office (PIO), Building 3, Room 3128, Mail Stop 522, 345 Middlefield Road, Menlo Park, CA 94025, (415) 329-4390.

____Geological Survey, National Cartographic Inforination Center (NCIC) California State Affiliate: Map and Imagery Laboratory Library, University of California, Santa Barbara, CA 93106, (805) 961-2779.

____Geological Survey, National Cartographic Information Center (NCIC) Idaho State Affiliate: Idaho State Historical Library. 610 N. Julia Davis Dr., Boise, ID 83702, (208) 334-3356.

____Geological Survey, National Cartographic Information Center (NCIC) Oregon State Affiliate: Oregon State Library, Public Services, Salem, OR 97310 (503) 378-4368.

____Geological Survey, National Cartographic Information Center (NCIC) Washington State Affiliate: Washington State Library, Information Services Division, Olympia, WA 98504, (206) 753-4027.

____Geological Survey, Water Resources Division District Office (California), Federal Building, Room W-2235, 2800 Cottage Way, Sacramento, CA 95825, (916) 978-4633.

____Geological Survey, Water Resources Division District Office (Idaho), 230 Collins Road, Boise, ID 83702, (208) 334-1750.

____Geological Survey, Water Resources Division District Office (Oregon), 847 NE 19th Avenue, Suite 300, Portland, OR 97232, (503) 231-2009.

____Geological Survey, Water Resources Division District Office (Washington), 1201 Pacific Avenue, Suite 600, Tacoma, WA 98402, (206) 593-6510.

U.S. Environmental Protection Agency.

____Region IX, 75 Hawthorne St., San Francisco, CA (415) 744-1305.

____Region X, 1200 Sixth Ave., Seattle, WA 98101, (206) 553-1200.

____Idaho Office, 1435 N. Orchard St., Boise, ID 83706 (208) 378-5746.

____Oregon Office, 811 SW Sixth Ave, Portland, OR 97204 (503) 326 3250.

____Washington Office, 300 Desmond Dr. SE, Lacey, WA (360) 753-9437.

A.4.2 State Offices

California

California Department of Fish and Game.

____Headquarters. 1416 Ninth St. Sacramento, CA 95814, (916) 653-7664. For further information about the Geographic Inforination System contact: John Ellison, 1730 I Street, Suite 100 Sacramento, CA 95814, (916) 323-1477, email to jellison@dfg.ca.gov.

____Region I, Northern California–North Coast, 601 Locust St., Redding, CA 96001, (916) 225-2300.

____Region 2, Sacrainento Valley–Central Sierra, 1701 Nimbus Dr., Rancho Cordova, CA 95670, (916) 358-2900.

____Region 3, Central Coast, P.O. Box 47, Yountville, CA 94599, (707) 944-5500

____Region 4, San Joaquin Valley–Southern Sierra, 1234 Shaw Ave., Fresno, CA 93710, (209) 222-3761.

____Region 5, Southern California–Eastern Sierra, 330 Golden Shore, Suite 50, Long Beach, CA 90802, (310) 590-5132.

California Department of Forestry.

____Administrative Unit, 1416 Ninth St., Sacramento, CA 94244-2460, (916) 653-5121.

——Coast–Cascade Region, 135 Ridgway Ave, Santa Rosa, CA 95402, (707) 576-2275.

——Sierra–South Region, 1234 East Shaw Ave., Fresno, CA 93710, (209) 222-3714.

California Department of Water Resources.

——Headquarters, P.O. Box 942836, Sacramento, CA 94236-0001; or 1020-9th Street, Sacramento, CA 95814; Division Chief (916) 327-1646.

——Division of Local Assistance, Northern District, 2440 Main Street, Red Bluff, CA 96080-2398; District Chief (916) 529-7342.

——Division of Local Assistance, Central District 3251 S Street, Sacramento. CA 95816-7017; District Chief (916) 445-5631.

California Regional Water Quality Control Board.

——North Coast Region (1), 5550 Skyline Blvd., Suite A, Santa Rosa, CA 95403, (707) 576-2220, FAX (707) 523-0135.

——San Francisco Bay Region (2), 2102 Webster St., Suite 500, Oakland, CA 94612, (510) 286-1255, FAX (510) 286-1380.

——Central Coast Region (3). 81 Higuera St., Suite 200, San Luis Obispo, CA 93401-5427, (805) 549-3147, FAX (805) 543-0397.

——Central Valley Region (5S), 3443 Routier Rd., Suite A, Sacramento, CA 95827-3098, (916) 255-3000, FAX (916) 255-3015.

——Central Valley Region (5F), 3614 East Ashlan Ave., Fresno, CA 93726, (209) 445-5116, FAX (209) 445-5910.

——Central Valley Region–Redding Office (5R), 415 Knollcrest Dr., Redding, CA 96002, (916) 224-4845, FAX (916) 224-4857.

——Lahontan Region (6SLT), 2092 South Lake Talioe Blvd., Suite 2, South Lake, Talioe, CA 96150. (916) 542-5400, FAX (916) 544-2271.

Idaho

Idaho Department of Fish and Game.

——Headquarters, 600 S. Walnut, P.O. Box 25, Boise, ID 83707, (208) 334-3700.

——Panhandle Region, 2750 Kathleen Avenue, Coeur d'Alene, ID 83814. (208) 769-1414.

——Clearwater Region, 1540 Warner Avenue, Lewiston, ID 83501, (208) 799-5010.

——Southwest Region, 3101 S. Powerline Road, Nampa, ID 83686, (208) 465-8465 (from Boise call 887-6729).

——McCall Region, 555 Deinhard Lane, McCall, ID 83638, (208) 634-8137.

——Magic Valley Region, 868 East Main Street, P.O. Box 428, Jerome, ID 83338, (208) 324-4350.

——Southeast Region, 1345 Barton Road, Pocatello, ID 83204, (208) 232-4703.

——Upper Snake Region, 1515 Lincoln Road, Idaho Falls, ID 83401, (208) 525-7290.

——Salmon Region, 1214 Hwy 93 N., P.O. Box 1336, Salmon, ID 83467, (208) 756-2271.

Idaho Department of Lands.

——Coeur d'Alene Staff Headquarters, 701 River Ave., P.O. Box 670, Coeur d'Alene, ID 83816, (208) 769-1525.

——Priest Lake Area Office, Cavanaugh Bay #132, Coolin ID 83821, (208) 443-2516.

——Pend Preille Lake Area Office, P.O. Box 909, Sandpoint, ID 83864, (208) 263-5104.

——St. Joe Area Office, 1806 Main Ave., St. Maries, ID 83861, (208) 245-4551.

——Clearwater Area Office, 10230 Highway 12, Orofino, ID 83544, (208) 476-4587.

——Payette Lakes Area Office, 555 Deinhard Lane, McCall, ID 83638.

——Southwest Idaho Area Office, 8355 W. State St. Boise, ID 83703, (208) 334-3488.

——South Central Idaho Area Office, P.O. Box 149, Gooding, ID 83330, (208) 934-5606.

——Eastern Idaho Area Office, 3563 Ririe Highway, Idaho Falls, ID 83401, (208) 523-5398.

Idaho Department of Water Resources.

——Main office, 1301 North Orchard Road, Boise, ID, (208) 327-7900, FAX (208) 327-7866.

——Northern Region, 1910 Northwest Blvd., Suite 210, Coeur d'Alene, ID 83814-2615, (208) 769-1450, FAX (208) 769-1454.

——Western Region, 2735 Airport Way, Boise, ID 83705-5082, (208) 334-2190, FAX (208) 334-2348.

——Southern Region: 222 Shoshone St. East, Twin Falls, ID 83301-6105, (208) 736-3033, FAX (208) 736-3037.

——Eastern Region: 900 North Skyline Drive, Idaho Falls, ID 83402-6105, (208) 525-7161, FAX (208) 525-7177.

Idaho Division of Environmental Quality.

——Main office, 1410 Hilton, Boise, ID 83702, (208) 334-4250.

Oregon

Oregon Climate Service.

——Strand Ag Hall, Room 316, Oregon State University, Corvallis OR, 97331-2209. (541) 737-5705, FAX (541) 737-2540, email oregon@ats.orst.edu.

Oregon Department of Environmental Quality.

——Main office, 811 SW Sixth Ave, Portland, OR 97204, (503) 229-6121.

Oregon Department of Fish and Wildlife.

——Main office, 2501 SW First Ave., PO Box 59; Portland, OR 97207; General Phone Number, (503) 229-5406; General Information, (503) 229-5222; Habitat Conservation Division, (503) 229-6967; Wildlife Division, (503) 229-5454.

——Northwest Region, 7118 Vandenberg Ave., Corvallis, OR 97330, (541) 757-4186.

——Southwest Region, 4192 N Umpqua Hwy., Roseburg, OR 97470, (541) 440-3353.

——Central Region, 61374 Parrell Rd., Bend, OR 97702. (503) 388-6363.

——Northeast Region, 107 20th Street, LaGrande, OR, 97850. (503) 963-2138.

——Southeast Region, 237 S. Hines Blvd., P.O. Box 8, Hines. OR 97738, (503) 573-6582.

——Marine Region, Marine Science Drive, Bldg. 3, Newport, OR 97365, (541) 867-4741.

——Columbia Region, 17330 SE Evelyn St., Clackamas, OR 97015, (503) 657-2000.

Oregon Department of Forestry.

——Main office, 2600 State Street, Salem, OR 97310, (503) 945-7200; State Forester (503) 945-7211; Deputy State Forester (503) 945-7202; Assistant State Forester (503) 945-7205.

——Northwest Oregon Area Office, Area Director, 801 Gales Creek Road, Forest Grove, Oregon 97116-1199, (503) 357-2191, FAX (503) 357-4548.

——Northwest Oregon Area, Forest Grove District, District Forester, 801 Gales Creek Road, Forest Grove, Oregon 97116-1199, (503) 357-2191, FAX (503) 357-4548.

——Northwest Oregon Area, Tillainook District, District Forester, 4907 E. Third Street, Tillamook, Oregon 97141-2999, (503) 842-2545, FAX (503) 842-3143.

——Northwest Oregon Area, Astoria District, District Forester, Route 1, Box 950, Astoria, Oregon 97103, (503) 325-5451, FAX (503) 325-2756.

——Northwest Oregon Area, Clackamas-Marion District, District Forester, 14995 S. Hwy. 211, Molalla, Oregon 97038, (503) 829-2216, FAX (503) 829-4736.

——Northwest Oregon Area, West Oregon District, District Forester, 24533 Alsea Hwy., Philomath, Oregon 97370, (541) 929-3266, FAX (541) 929-5549.

——Northwest Oregon Area, South Ford District, Administrative Supervisor, 48300 Wilson River Hwy., Tillainook, Oregon 97141, (503) 842-8439, FAX (503) 842-6572.

——Southern Oregon Area Office, Area Director, 1758 N.E. Airport Road, Roseburg, Oregon 97470-1499, (541) 440-3412, FAX (541) 440-3424.

——Southern Oregon Area, Southwest Oregon District, District Forester, 5286 Table Rock Road, Central Point, Oregon 97502, (541) 664-3328, FAX (541) 776-6260.

——Southern Oregon Area, Coos District, District Forester, 300 Fifth Street, Bay Park, Coos Bay, Oregon 97420, (541) 267-4136, FAX (541) 269-2027.

——Southern Oregon Area, Western Lane District, District Forester, P.O. **Box** 157, Veneta, Oregon 97487-0157, (541) 935-2283, FAX (541) 935-0731.

——Southern Oregon Area, Eastern Lane District, District Forester, 3150 Main Street, Springfield, Oregon 97478, (541) 726-3588, FAX (541) 726-2501.

——Southern Oregon Area, Linn District, District Forester, 4690 Highway 20, Sweet Home, Oregon 97386, (541) 367-6108, FAX (541) 367-5613.

——Eastern Oregon Area Office, Area Director, 3501 E. 3rd. Street, Prineville, Oregon 97754, (503) 447-5658, FAX (503) 447-1469.

——Eastern Oregon Area, Northeast Oregon District, District Forester, 611 20th Street, La Grande, Oregon 97850, (503) 963-3168; FAX (503) 963-4832.

——Eastern Oregon Area, Central Oregon District, District Forester, 220710 Ochoco Hwy., Prineville, Oregon 97754, (503) 447-5658, FAX (503) 447-1469.

——Eastern Oregon Area, Klamath-Lake District, District Forester, 3400 Greensprings Drive, Klamath Falls, Oregon 97601, (541) 883-5681, FAX (541) 883-5555.

Land Conservation and Development Deparfment.

___ 1175 NE Court St., Salem, OR 97310, (503) 373-0050.

Washington

Washington Department of Fish and Wildlife.

___Region I, 8702 N. Division St., Spokane, WA 99218, (509)456-4082.

___Region I, 8702 N. Division St., Spokane, WA 99218, (509) 456-4082.

___Region 2, 1550 Alder St. N.W., Ephrata, WA 98823, (509) 754-4624.

___Region 3, 1701 S. 24th Ave., Yakima, WA 98902, (509) 575-2740.

___Region 4. 16018 Mill Creek Blvd., Mill Creek, WA 98012, (206) 775-1311.

___Region 5, 5405 NE Hazel Dell, Vancouver, WA 98663, (360) 696-6211.

___Region 6, 48 Devonshire Rd., Montesano, WA 98563, (360) 586-6129.

Washingfon Deparfmenf of Ecology

___Main office, P.O. Box 47600, Olympia, WA 98504, (360) 407-6000.

___Central Regional Office, 15 West Yakinia Ave., Suite 200, Yakima, WA 98902-3401, (509) 575-2490, FAX (509) 575-2809.

___Eastern Regional Office, N. 4601 Monroe, Suite 100, Spokane WA 99205-1295, (509) 456-2926, FAX (509) 456-6175.

___Northwest Regional Office, 3190-160th Ave. S.E., Bellevue, WA 98008-5452, (206) 649-7000, FAX (206) 649-7098.

___Southwest Regional Office, P.O. Box 47775, Olympia, WA 98504-7775, (360) 407-6300, FAX (360) 407-6305. Washington Department of Natural Resources.

___Habitat Conservation Planning Team, 1111 Washington St. S.E., MS-47011, Olympia, WA 98504-7011, (360) 902-1481, FAX 360-902-1790.

___Photo & Map Sales, P.O. Box 47013, Olympia, WA 98504-7013, (360) 902-1234.

Internet Sources

California Department of Water Resources (DWR) at **http://wwwdrw.water.ca.gov.**

California Department of Fish and Game, Natural Diversity Data Base (NDDB) at http://spock.dfg.ca.gov/Endangered/ endangered.html.

Idaho Department of Fish and Game at http ://www .state.id.us/fishgame/fishgame.html.

Idaho Department of Water Resources at http://www .state.id.us/idwr/idwrhome.html.

Oregon Department of Fish and Wildlife (ODFW), homepage at http://www.dfw.state.or.us. Oregon Rivers Inforination System (ORIS), Northwest Environmental Database, can be accessed through Brent O. Forsberg, Coordinator at forsberg@dfw.or.gov. or the ODFW homepage.

U.S. Department of Agriculture.
___ National Resource Conservation Service, National Resources Inventory (NRI) at http://www.ncg.nrcs. usda.gov/nri.html.

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99 Stat 1354. 1985. Food Security Act (FSA). Passed as PL 99-198, 23 December 1985, Title XII--Conservation.

Abstract

Populations of wild anadromous and resident salmonids continue to decline throughout much of the Pacific Northwest and northern California. Several stocks are presently listed as threatened or endangered under the Federal Endangered Species Act. Degradation of freshwater and estuarine habitats contribute substantially to this decline. Although Federal, State, and Tribal programs have been established, no coordinated, region-wide strategy exists to develop habitat conservation plans, foster habitat protection and restoration beyond minimum requirements on nonfederal lands, or encourage education and training.

This document provides the technical basis from which government agencies and landowners can develop and implement an ecosystem approach to habitat conservation planning, protection, and restoration of aquatic habitat on nonfederal lands. The report also describes a process for developing, approving, and monitoring habitat conservation plans, pre-listing agreements, and other conservation agreements for nonfederal lands to be consistent with the mandates of applicable legal requirements. Three parts constitute the body of the document. Chapters 1–10 supply the technical foundation for understanding salmonid conservation principles from an ecosystem perspective: over 50 years of reported scientific research has been synthesized to describe physical, chemical, and biological processes operating across the landscape, within riparian areas, and in aquatic ecosystems as well as the effects of human activities on these processes. Chapters 11–16 provide a general conceptual framework for achieving salmonid conservation on nonfederal lands in the Pacific Northwest, including specific guidelines for developing, monitoring, and implementing habitat conservation plans within the larger context of basin and regional conservation goals. An appendix lists information resources that landowners and agencies may find useful in developing and evaluating habitat conservation plans. Over 1100 sources are cited within this document.

The perspective we present in this document is anchored in the natural sciences. Although we touch on social, economic, and ethical concerns, an exhaustive discussion of these issues is beyond the report's scope. Nevertheless, our socioeconomic systems and values shape our perceptions of natural resources and drive our demands for them. The fate of salmonids in the Pacific Northwest is inextricably interwoven into this natural-cultural fabric. Just as conservation strategies that are not based on sound ecological principles will ultimately fail, ecological approaches that ignore socioeconomic values, political realities, and ethical issues are also at high risk of failure. In light of this inter-dependency between biological and social realms, we view this document as one piece of a conservation-restoration puzzle to be integrated into a more comprehensive assessment of what we as a society want and value, what legacy we wish leave to future generations, and how we can get there from here.

Key words

salmonids, aquatic ecosystems, aquatic habitat, land-use effects, environmental monitoring, environmental law, environmental regulations, disturbances, management systems, riparian habitat, watershed processes, habitat restoration, conservation

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Preface

Populations of wild anadromous and resident salmonids are in decline throughout much of the Pacific Northwest and northern California. Several stocks are presently listed as threatened or endangered under the Federal Endangered Species Act (ESA), and continued losses are likely to result in additional ESA listings. A significant cause of salmonid declines is degradation of their freshwater and estuarine habitats. Although Federal, State, and Tribal conservation and restoration programs have been established, there is no coordinated, region-wide Federal strategy for developing habitat conservation plans pursuant to ESA, for fostering habitat protection and restoration beyond minimum ESA requirements on nonfederal lands, or for providing education and training in habitat protection and restoration strategies.

The National Marine Fisheries Service, the Environmental Protection Agency, and the Fish and Wildlife Service (the "Agencies") seek to develop 1) a training and outreach strategy to implement a coordinated ecosystem approach to ESA's habitat conservation planning as well as additional protection and restoration of aquatic habitat on nonfederal lands and 2) a process for developing, approving, and monitoring habitat conservation plans (HCPs), pre-listing agreements, and other conservation agreements for nonfederal lands that is consistent with the mandates of ESA, the Clean Water Act, and other applicable State and Federal requirements. This document provides the technical basis from which these goals can be accomplished. The primary intended audience is agency personnel who have background in the biological and physical sciences and who are responsible for overseeing land management activities. Use of technical terms that may be unfamiliar to some readers was at times unavoidable; consequently, the document may be less accessible to those without formal technical training in scientific disciplines.

The document is organized generally into three parts. Chapters 1–10 (Part I) provide the technical foundation for understanding salmonid conservation principles from an ecosystem perspective. We discuss the physical, chemical, and biological processes operating across the landscape, within riparian areas, and in aquatic ecosystems; these processes ultimately influence the ability of streams, rivers, and estuaries to support salmonids. Specific habitat requirements of salmonids during each life stage are detailed. We then review the effects of land-use practices on watershed processes and salmonid habitats, focusing on the impacts of logging, grazing, farming, mining, and urbanization on hydrology, sediment delivery, channel morphology, stream temperatures, and riparian function. An overview is presented on the importance of ocean variability in determining production of anadromous salmonids and the implications of this variability on restoration of freshwater habitats of salmonids. Next, land-use practices that minimize impacts to salmonids and their habitats are discussed, followed by a brief review of Federal laws that pertain to the conservation of salmonids on private lands. The Technical Foundation concludes with a review of strengths and weaknesses of existing programs for monitoring aquatic ecosystems; this chapter provides the basis for monitoring recommendations presented in Part II.

Chapters 11–16 (Part II) provide a general conceptual framework for achieving salmonid conservation on nonfederal lands in the Pacific Northwest, as well as specific guidelines for the development of Habitat Conservation Plans (HCPs) pursuant to the Endangered Species Act. We propose a hierarchical approach to the development and evaluation of HCPs and other conservation efforts, stressing the need for site- or watershed-level conservation efforts to be developed and evaluated within the larger context of basin and regional conservation goals. We outline critical issues that should be addressed at the scales of region and basin, watersheds, and individual sites while planning HCPs. We present details of specific elements for planning effective HCPs and criteria for evaluating the potential effectiveness of HCP provisions where such criteria are supported by current scientific information. Included in this discussion is an evaluation of the effectiveness of State rules for riparian management to protect specific processes that directly affect aquatic habitats. Compliance and assessment monitoring strategies for HCPs and other conservation efforts are proposed. The document concludes with a suggested strategy for implementing

salmonid conservation efforts on nonfederal lands. An appendix (the third part) lists sources of data that landowners and agencies may find useful in developing and evaluating habitat conservation plans. Over 1100 sources are cited within this document and listed in the references section.

The perspective we present in this document found its anchor in the natural sciences. Although we touch on social, economic, and ethical concerns, an exhaustive discussion of these issues is beyond the scope of the document. Nevertheless, it is our socio-economic systems and values that shape our perceptions of natural resources and drive our demands for them. The fate of salmonids in the Pacific Northwest is inextricably interwoven into this natural-cultural fabric. Just as conservation strategies that are not based on sound ecological principles will ultimately fail, ecological approaches that ignore socioeconomic values, political realities, and ethical issues are also at high risk of failure. Scientific information influences how society both views and values natural resources such as salmon. At the same time, social values influence where we devote our research efforts (and hence the strengths and weaknesses of our knowledge base) and the feasibility of implementing what is ecologically sound. In light of this interdependency between the biological and social realms, we view this document as one piece of a conservation and restoration puzzle to be integrated into a more comprehensive assessment of what we as a society want and value, what legacy we wish leave to future generations, and how we can get there from here.

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Finally, numerous others—too many to mention individually—have met with us to discus specific issues, have suggested documents to review or other experts to contact, have provided us materials to review, or have offered access to libraries and document collections.

Acronyms

AFS	American Fisheries Society
BIA	Bureau of Indian Affairs
BLM	Bureau of Land Management (DOI)
BOR	Bureau of Reclamation (DOI)
CDF	California Department of Forestry
CDFG	California Department of Fish & Game
CDWR	California Department of Water Resources
CFR	Code of Federal Regulations
CRFC	Columbia River Fisheries Council
CSWQCB	California State Water Quality Control Board
DO	dissolved oxygen
DOC	U.S. Department of Commerce
DOI	U.S. Department of Interior
EMAP	Environmental Monitoring and Assessment Program (EPA)
EPA	U.S. Environmental Protection Agency
ERS	Economic Research Service (USDA)
FEMAT	Forest Ecosystem Management Assessment Team
FS	Forest Service (USDA)
FWS	Fish and Wildlife Service (DOI)
GAO	General Accounting Office (U.S. Congress)
GIS	geographic information system
GPO	Government Printing Office
GS	Geologic Survey (DOI)
IDFG	Idaho Department of Fish & Game
IDWR	Idaho Department of Water Resources
LWD	large woody debris
MSG	California State Board of Forestry Monitoring Study Group
NIFC	Northwest Intertribal Fish Commission
NMFS	National Marine Fisheries Service (DOC, NOAA)
NOAA	National Oceanic and Atmospheric Administration (DOC)
NPCC	Northwest Power Planning Council
NPS	National Park Service (DOI)
NRC	National Research Council
NRCS	Natural Resources Conservation Service {formerly SCS} (USDA)
ODEQ	Oregon Department of Environmental Quality
ODF	Oregon Department of Forestry
ODFW	Oregon Department of Fish and Wildlife
ODSL	Oregon Division of State Lands
OSPRD	Oregon State Parks & Recreation Department
OWRRI	Oregon Water Resources Research Institute
PAH	polyaromatic hydrocarbon
PCB	polychlorinated biphenyl
PFMC	Pacific Fishery Management Council
PSMFC	Pacific States Marine Fisheries Commission
PSWQA	Puget Sound Water Quality Authority
PWD	Public Works Department, City of Olympia, Washington
REO	Regional Ecosystem Office [BLM, NMFS, FWS, BIA, EPA, FS]
SAB	Science Advisory Board {for EPA)
SCS	Soil Conservation Service (superseded by NRCS} (USDA)
TFW	Timber, Fish and Wildlife Group
TU	turbidity units
URL	universal resource locator
USC	U.S. Code
USDA	U.S. Department of Agriculture
WACT	Watershed Analysis Coordination Team
WDE	Washington Department of Ecology
WDF	Washington Department of Fisheries (superseded by WDFW)
WDFW	Washington Department of Fisheries and Wildlife {superseded WDF and WDW)
WDNR	Washington Department of Natural Resources
WDW	Washington Department of Wildlife (superseded by WDFW}
WFPB	Washington Forest Practices Board
WSSP	Washington State Shoreline Program

Parentheses () indicate the department to which an agency belongs.
Brackets [ ] indicate member agencies.
Squiggly brackets (} indicate additional information.

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1 Executive Summary: Part I

1.1 Introduction

As substantial evidence accumulates, concerns grow amid continuing declines of salmonids in the Pacific Northwest. Anadromous salmonids returning to the Columbia River to spawn have decreased from historical highs of 10–16 million wild fish to fewer than 2 million fish, mostly originating from hatcheries. At least 106 wild salmon stocks have been extirpated, 214 are at high or moderate risk of extinction, and many have been listed or are being reviewed for listing under the Endangered Species Act. Similarly, several resident species and stocks have also been proposed for listing. Salmon fisheries along coastal regions of Oregon and California have been dramatically curtailed because of dwindling numbers of fish and increasing concern for wild stocks. A number of natural and anthropogenic factors have contributed to these declines: hydropower operations, over exploitation, artificial propagation, climatic and oceanic changes, and destruction and degradation of habitat through land-use and water-use practices. Although the relative impact of these different factors varies among basins and river systems, habitat loss and degradation are considered contributing factors in the decline of most salmonid populations.

Part I of *An Ecosystem Approach to Salmonid Conservation* is intended to provide a comprehensive technical foundation for understanding salmonid conservation principles in an ecosystem context. Aquatic habitats critical to salmonids are the product of processes acting throughout watersheds and particularly within riparian areas along streams and rivers. This document depends on the premise that salmonid conservation can be achieved only by maintaining and restoring these processes and their natural rates. If ecosystems are allowed to function in a natural manner, habitat characteristics favorable to salmonids will result, and fish will be able to reinvade and populate historical habitats, recover from earlier stressors, and persist under natural disturbance regimes. This ecosystem-oriented approach complements recent Federal and State strategies that emphasize watershed and landscape-level functions of ecosystems for management and conservation of forest resources.

After briefly reviewing evidence of trends for Pacific Northwest salmonids (Chapter 2), we discuss physical, chemical, and biological processes that affect aquatic ecosystems and the salmonids that inhabit them (Chapters 3 and 4). Next, we present an overview of habitat requirements of salmonids, including elements that are essential to the general health of aquatic ecosystems, as well as specific habitat requirements at each life stage of salmonids (Chapter 5). We then discuss how human activities affect watershed and instream processes, focusing on effects of logging, grazing, agriculture (including irrigation withdrawal), mining, and urbanization (Chapter 6). Effects of dams, species introductions (including hatchery practices), and salmon harvest are presented more briefly since these topics, while important in providing context for the document, were beyond the scope of this project. We also review the influence of climatic and oceanic conditions on salmonids and how these factors relate to salmonid conservation (Chapter 7). Next, we present an overview of management practices and programs that reduce the detrimental effects of human activities on salmonids (Chapter 8), followed by a discussion of Federal laws and regulations relevant to the conservation of salmonids (Chapter 9). Part I concludes with a review of strengths and weaknesses of existing monitoring programs for aquatic ecosystems (Chapter 10). In Part II of this document, we provide a general conceptual framework for achieving salmonid conservation on nonfederal lands in the Pacific Northwest as well as specific guidelines for the development of salmonid conservation plans, including Habitat Conservation Plans (HCPs), prepared pursuant to the Endangered Species Act. A separate executive summary (Chapter 11) describes major findings and recommendations related to conservation planning (Chapter 12–15).

This document focuses on anadromous salmonid species, including five Pacific salmon (chinook, coho, chum, pink, and sockeye), trout and char with both resident and anadromous forms (rainbow, cutthroat, and bull trout), and strictly resident species (mountain whitefish). The areal scope was limited to the portions of the States of California, Idaho, Oregon, and Washington that have supported salmonid populations. For many subject areas, we have relied heavily on comprehensive literature reviews and syntheses already available in the scientific literature. For subject areas where no such

summaries were available, we have conducted more extensive literature reviews.

1.2 Physical and Chemical Processes

The physical and chemical characteristics of streams, rivers, lakes, and estuaries of the Pacific Northwest are the manifestation of processes operating at many temporal and spatial scales. Tectonic activity and glaciation have continually reshaped the landscape of the Pacific Northwest over millions of years. Alternating glacial and interglacial periods have caused changes in vegetative cover and geomorphic processes over significant portions of the region. Present climatic conditions have prevailed for the past 6,000 to 8,000 years, and modern coniferous forest communities developed over much of the coastal region within the last 2,000 to 5,000 years. In response to these changes, many river channels have shifted from unstable braided channels to relatively stable, meandering channels because the relative influence of hydrology, sediment delivery, and woody debris have changed.

Over periods of decades to centuries, large floods, fires, and mass wastings have been dominant natural disturbances influencing river channels. These disturbances can cause abrupt changes in habitat conditions, reconfiguring the stream channel, transporting streambed materials, depositing large quantities of coarse and fine sediments to streams, and altering hydrologic and nutrient cycling processes. These changes may persist for decades or more, affecting the relative suitability of habitats to various salmonids.

At the watershed and site levels, the major processes that affect the physical and chemical attributes of aquatic ecosystems are hydrology, sediment transport, heat energy transfer, nutrient cycling/solute transport, and delivery of large woody debris to streams. Runoff from the watershed affects stream habitats directly by determining the timing and quantity of streamflow, which control habitat availability and influence channel configuration, and indirectly by affecting the processes of energy transfer, sediment transfer, and nutrient cycling/solute transport. The amount of water reaching streams is a function of precipitation patterns, evapotranspiration losses, and infiltration rates, which in turn are affected by watershed characteristics including local climate, topography, soil type, slope, and vegetative cover. Hydrologic regimes of streams in the Pacific Northwest can be divided into three general patterns: rain-dominated systems, which are hydrologically flashy because of frequent rainstorms during the winter (coastal mountains, lowland valleys, and lower elevations of the Cascade and Sierra Nevada Mountains); transient-snow systems, which exhibit both rain and snow during the winter and may experience high flows associated with rain-on-snow events (mid-elevation of the Cascade, northern Sierra Nevada, and Olympic Mountains); and snow-dominated systems, where most precipitation falls as snow during the winter months and is delivered to streams in the spring as snow melts (higher elevations of the Cascade, Sierra Nevada, Olympic, and Rocky Mountains, and mid-elevation areas east of the Cascade/Sierra Crest).

Sediment from upland and riparian areas plays a major role in determining the nature and quality of salmonid habitats in streams, rivers, and estuaries. Sediment is generated from surface erosion and mass-wasting processes. Surface erosion occurs when soil particles are detached by wind, rain, overland flow, freeze-thaw, or other disturbance (animals, machinery) and transported to the stream channel. Mass wasting (slumps, earthflows, landslides, debris avalanches, and soil creep) results from weathering, freeze-thaw, soil saturation, groundwater flow, earthquakes, undercutting of streambanks, and wind stress transferred to soil by trees. Bank erosion and bedload movement occur naturally during high flows, but both may be exacerbated where riparian vegetation that stabilized banks is removed or when peak flows are increased by human activities. Watershed characteristics affecting sediment transport include climate, topography, geology, soil type and erodibility, vegetative cover, and riparian zone characteristics. West of the Cascades, mass wasting is the major source of sediments in undisturbed systems; east of the Cascades, both surface erosion and mass wasting may be important sources of sediments. In general, rain-dominated systems tend to yield more sediment that snow-dominated systems, although interbasin variability is high because of differences in topography, total precipitation, and soil type.

Stream temperatures influence virtually all aspects of salmonid biology and ecology, affecting the development, physiology, and behavior of fish, as well as mediating competitive, predator-prey, and disease-host relationships. Heat energy is transferred to streams and rivers by six processes: short-wave radiation (primarily solar), long-wave radiation, convective mixing with the air, evaporation, conduction with the stream bed, and advective mixing with inflow from ground water or tributaries. The temperature of streams represents a balancing of these factors. During the summer, incoming solar radiation is the dominant source of energy for smaller streams, though groundwater discharge may be locally important. Consequently, riparian vegetation plays a major role in controlling summer stream temperatures as may topographic features that provide shade. During the winter, direct solar radiation becomes less important because of lower

sun angles, shorter days, and cloudier conditions. Stream characteristics, including width, depth, velocity, and substrate also determine the rate at which heat is gained or lost through radiation, convection, conduction, and evaporation. As streams become larger and less shaded downstream, the influence of both terrestrial vegetation and groundwater inputs diminishes, and temperatures tend to equilibrate with mean air temperatures.

Water is the primary agent dissolving and transporting solutes and particulate matter across the landscape, integrating processes of chemical delivery in precipitation, weathering, erosion, chemical exchange, physical adsorption and absorption, and biotic uptake and release. Climate, geology, and biological processes all influence the character and availability of inorganic solutes. The composition and age of parent rock determine the rate of weathering and hence the release of soluble materials. These dissolved materials are transported by surface and groundwater flow to streams. The biota of terrestrial, riparian, and aquatic ecosystems mediate the sources and cycling of major nutrients and associated organic solutes through processes such as photosynthesis, respiration, food uptake, migration, litter fall, and physical retention. Side channels on floodplains are areas of high nutrient uptake and processing because of low current velocities and extensive contact with the water column. Riparian vegetation may remove a significant proportion of the available phosphorous and nitrogen (60%–90%)and thus directly affects stream productivity.

Once in the stream, nutrients are transported downstream until they are taken up and processed by organisms and then released again, collectively termed "nutrient spiraling." The average distance over which one complete spiral occurs varies with stream characteristics, including retentive structures that physically trap particulate matter, stream size, water velocity, and the degree of contact between the water column and biological organisms inhabiting the stream bed. Simplification of channel structure increases nutrient spiral length, decreasing retention efficiency. Salmon and lamprey carcasses are also an integral part of nutrient cycling for both aquatic and riparian systems; thus declines in salmonids may cause more fundamental changes in ecosystem productivity than the simple loss of stocks or species.

Riparian and floodplain areas are the critical interface between terrestrial and aquatic ecosystems, serving to filter, retain, and process materials in transit from uplands to streams. Riparian vegetation plays a major role in providing shade to streams and overhanging cover used by salmonids. Streamside vegetation stabilizes stream banks by providing root mass to maintain bank integrity, by producing hydraulic roughness to slow water velocities, and by promoting bank building through retention of sediments. Riparian vegetation also provides much of the organic litter required to support biotic activity within the stream as well as the large woody debris needed to create physical structure, develop pool-riffle characteristics, retain gravels and organic litter, provide substrate for aquatic invertebrates, moderate flood disturbances, and provide refugia for organisms during floods. Large woody debris performs important functions in streams, increasing channel complexity, creating hydraulic heterogeneity, and providing cover for fish. Large wood also provides critical habitat heterogeneity and cover in lakes, estuaries, and the ocean. In addition to the aquatic functions that riparian areas perform, they typically provide habitat and create unique microclimates important to a majority of the wildlife occupying the watershed.

1.3 Biological Processes

The physiology and behavior of organisms, the dynamics and evolution of populations, and the trophic structure of aquatic communities are influenced by the spatial and temporal patterns of water quantity and velocity, temperature, substrate, physical structure, and dissolved materials. At the organism level, survival of salmonids depends on their ability to carry out basic biological and physiological functions including feeding, growth, respiration, smoltification, migration, and reproduction. All of the habitat characteristics listed above influence the quality and amount of food energy available, the amount of energy expended for metabolic processes, and hence the amount available for growth, migration, and reproduction.

Each phase of the salmonid life cycle—adult maturation and migration, spawning, incubation of embryos and alevins, emergence of fry, juvenile rearing, and smolt migration—may require utilization of and access to distinct habitats. The strong homing ability of salmonids has led to the formation of numerous, relatively isolated stocks, each adapted to the specific environmental conditions found in its natal and rearing habitats. This adaptation is reflected in the wide diversity of life histories exhibited by the salmonids of the Pacific Northwest. A major concern is that land use and water use have reduced habitat diversity through loss or simplification of habitat, which in turn has reduced the life-history diversity exhibited in the salmonid populations. At larger spatial scales, groups of populations or "metapopulations" interact infrequently through straying or dispersal. Metapopulation theory suggests local populations within metapopulations periodically go extinct and are recolonized and that metapopulations will persist if recolonization rates exceed extinction rates. The core-satellite

metapopulation model proposes that extinction probability is not equal among populations and that certain extinction-resistant populations are important "seed" sources of recolonizers for habitats made vacant by extinction. Conservation of salmonids thus depends on maintaining: connectivity among habitats to allow reinvasion of vacant habitats, sufficient genetic diversity to allow successful recolonization of these habitats, and refugia from which dispersal can occur. The concept of Evolutionarily Significant Units presently being used by Federal agencies to determine appropriate units of conservation for salmonids is based in part on these metapopulation considerations.

Biotic communities in aquatic ecosystems are influenced by predator-prey, competitive, and disease- or parasite-host relationships within and among species. Current theory suggests that disturbance plays a major role in influencing the outcome of these interactions and, thus, in determining community or assemblage structure. Two models appear applicable to stream communities. The "intermediate disturbance hypothesis" argues that diversity is greatest in systems experiencing intermediate disturbance, because neither colonizers (favored by frequent disturbance) nor superior competitors (favored by infrequent disturbance) can maintain dominance. The "dynamic equilibrium model" proposes that community structure is a function of growth rates, rates of competitive exclusion, and frequency of population reductions; inferior competitors persist if disturbance occurs often enough to prevent competitive exclusion, but species with long life cycles are lost if disturbance is too frequent. Both of these theories suggest that increases in disturbance frequency caused by human activities are likely to alter community structure.

Food webs in aquatic systems are highly complex, consisting of many species representing several trophic levels. These food webs can be highly modified by environmental changes in light energy or nutrient inputs; alterations of streamflow, temperature, or substrate; and introductions of non-native organisms. Changes in physical habitat characteristics can alter competitive interactions within and among species. Similarly, changes in temperature or flow regimes may favor species that prey on salmonids, such as northern squawfish and a host of introduced predators, Salmonids are affected by a variety of bacterial, viral, fungal, and microparasitic pathogens. Both the immune system of fishes and the virulence of pathogens are greatly affected by environmental conditions, especially temperature; thus, alteration of temperature, substrate, and flow may increase the incidence of epizootics.

1.4 Salmonid Habitat Requirements

Operating throughout the watershed and across the landscape, all of the physical, chemical, and biological processes discussed above affect the features and characteristics of aquatic habitats from headwater streams and lakes to estuaries and the ocean. To protect or restore desirable habitat requires that the natural processes producing those features and characteristics must be maintained or restored. Four general principles should be considered when determining habitat requirements of salmonids:

- Watersheds and streams differ in their flow, temperature, sedimentation, nutrients, physical structure, and biological components.
- Fish populations adapt and have adapted—biochemically, physiologically, morphologically, and behaviorally—to the natural environmental fluctuations that they experience and to the biota with which they share the stream, lake, or estuary.
- Specific habitat requirements of salmonids differ among species and life-history types, and these requirements change with season, life stage, and the presence of other biota.
- Aquatic ecosystems change over evolutionary time.

Consequently, there are no simple definitions of salmonid habitat requirements, and the goal of salmonid conservation should be to maintain habitat elements within the natural range for the particular system.

Five general classes of features or characteristics determine the suitability of aquatic habitats for salmonids: flow regime, water quality, habitat structure, food (energy) source, and biotic interactions. Flow regimes directly influence the depth and velocity of water and the total available habitat space for salmonids and their food organisms as well as perform other functions such as redistributing sediments, flushing gravels, and dispersing vegetation propagules. Water quality requirements include cool temperatures, high dissolved oxygen, natural nutrient concentrations, and low levels of pollutants. Salmonids prefer cold water, and temperatures above 25°C are lethal to most species; individual species have specific preference ranges that vary by life stage. Variation in temperature is required to trigger spawning, support growth, initiate smoltification, and enable other parts of the salmonid life cycle. Salmonids require well oxygenated water (> 6 mg/l) throughout their life cycles, and any level below saturation can be detrimental. Nutrient levels vary among streams and must be sufficient to support natural plant and animal assemblages, Important structural attributes of

streams include pools, riffles, substrate, cover (e.g., undercut banks, overhanging vegetation), depth, and hydraulic complexity. The presence of large woody debris enhances channel complexity, creating hydraulic heterogeneity, pools, side channels, back eddies, and other features that are used by salmonids and other aquatic organisms. Maintaining adequate food sources depends upon maintaining natural inputs of allochthonous material (type, amount, and timing) as well as physical structures needed to retain these materials. Normal biotic interactions also must be maintained to ensure the health of aquatic ecosystems, including competitive, predator-prey, and disease-parasite relations.

Stream habitat and channel features vary markedly from headwater streams to the estuaries and ocean. Salmonids, particularly anadromous species, use the entire range of habitats encountered during completion of their life cycles. The diversity of life histories exhibited by salmonids has developed to accommodate and fully exploit the range of habitats encountered. Loss of specific elements of habitat diversity may reduce the diversity exhibited in the salmonids' life histories, which in turn may influence the ability of these fish to adapt to natural and anthropogenic change.

Habitat requirements vary by life stage. During spawning migrations, adult salmon require water of high quality (cool temperatures or thermal refugia, dissolved oxygen near 100%, and low turbidity); adequate flows and depths to allow passage over barriers to reach spawning sites; and sufficient holding and resting sites. Spawning areas are selected on the basis of species-specific requirements of flow, water quality, substrate size, and groundwater upwelling. Embryo survival and fry emergence depend upon substrate conditions, including gravel size, porosity, permeability, and oxygen levels; substrate stability during high flows; and appropriate water temperatures ($<$ 14°C for most species, but $<$ 6°C for bull trout). Habitat requirements for rearing juveniles of anadromous species and adults of resident species also vary with species and size. Microhabitat requirements for holding, feeding, and resting each differ, and these requirements change with season. Migration of juveniles to rearing areas (whether the ocean, lakes, or other stream reaches) requires unobstructed access to these habitats. Physical, chemical, and thermal conditions may all impede migrations of juvenile fish.

1.5 Effects of Human Activities on Watershed Processes, Salmonids, and Their Habitats

Land-use practices, including forestry, grazing, agriculture, urbanization, and mining can substantially alter watershed processes, resulting in degradation of streams, lakes, and estuaries. Logging and grazing affect the greatest percentage of lands in the Pacific Northwest, but effects of agriculture, urbanization, and mining may result in a higher degree of local disturbance. Most of the alterations from land-use practices in upland areas result from changes in vegetation and soil characteristics, which in turn affect the quantity and routing of water, sediments, nutrients, and other dissolved materials delivered to streams. In addition, application of chemical fertilizers and biocides can affect water quality. Activities within the riparian zone can alter shading (and hence stream temperature), transport and supply of sediment, inputs of organic litter and large wood, bank stability, seasonal streamflow regimes, and flood dynamics. Dams, irrigation diversions, and road crossings hinder migrations, alter physical and chemical character of streams, and change the composition of stream biota. Harvest of salmonids reduces the abundance and alters the size-and-age structure of populations. Introduced fish species can adversely affect native salmonids through competition, predation, and disruption of physical habitat. Similarly, hatchery-reared salmonids may have similar impacts as well as altering the genetic structure of populations through introgression.

1.5.1 Foresfry

Forest practices result in removal and disturbance of natural vegetation, disturbance and compaction of soils, construction of roads, and installation of culverts. Removal of vegetation typically reduces water loss to evapotranspiration, resulting in increased water yield from the watershed. In general, increases in water yield are greater west of the Cascades than they are on the east side. Increases in peak flows following logging have been reported and likely result from combined effects of vegetation removal and more rapid routing of water from uplands to the stream channel. Short-term increases in summer base flows frequently follow logging; however, evidence from one Cascade watershed suggests base flows may be reduced over the long term, particularly if coniferous vegetation is replaced by hardwood- dominated stands.

Site disturbance and road construction typically increase sediment delivered to streams through mass wasting and surface erosion, which can elevate the level of fine sediments in spawning gravels and fill substrate interstices that provide habitat for aquatic invertebrates. The removal of riparian canopy reduces shading and increases the amount of solar radiation reaching the streams, resulting in higher maximum stream temperatures and increased diel and seasonal fluctuations. In addition, the loss of riparian vegetation may increase radiative cooling during the

winter, enhancing the formation of anchor ice. In other systems, increases in winter stream temperatures have been observed after logging. Increases in maximum temperature after logging depend on the size and morphology of the stream and on the type and density of canopy removed. Altered stream temperatures persist until prelogging levels of shade are re-established, which may take from less than 10 to more than 40 years.

Timber harvest removes plant biomass, and hence nutrients, but nutrients are more available to streams immediately following harvest, resulting in part from addition of slash to the forest floor, accelerated decomposition of litter, and increased runoff and erosion. This short-term increase diminishes as soils stabilize and revegetation occurs. Where logging occurs in riparian areas, delivery of leaf litter and large woody debris to the stream is reduced, and may significantly alter the nutrient balance and physical character of the stream. Loss of large woody debris, combined with alteration of hydrology and sediment transport, reduces complexity of stream micro- and macrohabitats and causes loss of pools and channel sinuosity. These alterations may persist from decades to centuries. Changes in habitat conditions may affect fish assemblage structure and diversity (e.g., favoring species that prefer riffles rather than pools), alter the age-structure of salmonid populations, and disrupt the timing of life-history events. Other effects on salmonids include reduced embryo survival and fry production, decreased growth efficiency, increased susceptibility to disease and predation, lower overwinter survival, blocked migration (e.g., inadequate culverts), and increased mortality through anglers' improved access to streams.

1.5.2 Grazing

Grazing results in the removal of natural vegetation, the alteration of plant-community composition, and the modification of soil characteristics, which in turn affect hydrologic and erosional processes. Effects are particularly acute in the riparian zone, where livestock tend to congregate, attracted by water, shade, cooler temperatures, and an abundance of high-quality forage. In general, grazed lands have less vegetation and litter cover than ungrazed lands, and in many areas of the West, perennial grasses have been replaced by non-native annual grasses and weedy species. Greater exposure of soils leads to splash erosion, which decreases soil permeability and results in more rapid runoff of precipitation to the stream channel. **As** a consequence, peak flows may be higher and summer base flows lower in watersheds that are intensively grazed.

Livestock also affect vegetation and soils through trampling. Trampling soils in arid and semi-arid lands may break up the fragile cryptogamic crust (comprised of symbiotic mosses, algae, and lichens) causing reduced infiltration, increased runoff, and reduced availability of nitrogen for plant growth. In addition, trampling detaches soil particles, accelerating surface erosion in upland areas, and may promote mass wasting along streambanks. Mass wasting also occurs where grazing has eliminated riparian vegetation and hence the root matrix that helps bind soil together. All of these processes result in increased sediment transport to streams. Animals also redistribute seeds and nutrients across the landscape, especially to riparian zones or other attractors, such as spring seeps or salt blocks. Devegetating riparian zones reduces shading and increases summer stream temperatures—often in streams that are where temperatures are near the upper limit of the tolerable range for salmonids— and may also increase the formation of anchor ice in the winter. Grazing also results in changes in channel morphology through changes in hydrology, sedimentation, and loss of bank stability. Streams in grazed areas tend to be wider and shallower, and consequently warmer in summer, than in ungrazed reaches. In some instances, streams in grazed areas incise in response to increased peak flows, effectively disconnecting the stream channel from the floodplain. Incision further alters the hydrology of the stream by lowering the water table and, consequently, the plant community occupying the riparian zone may shift from hydric (wetland) to xeric vegetation. Grazing in the riparian zone can reduce recruitment of large woody debris, especially because re-establishment of riparian shrubs and trees rarely occurs if grazing pressure is not reduced. Loss of woody debris reduces retention of gravels, creation and maintenance of pool habitats, and instream cover. General effects of grazing on salmonids include reduced reproductive success because of sedimentation of spawning gravels, alteration of food supplies through changes in primary and secondary production, reduced fish densities, and shifts in the composition of fish, invertebrate, and algal communities.

1.5.3 Agriculture

Although agriculture is not a dominant land use in the Pacific Northwest (approximately 16% of the total land area), alterations to the land surface are more severe than those caused by forestry or grazing, are generally permanent, and tend to involve repeated disturbance. Replacing natural grasslands, forests, and wetlands with annual crops leaves much area unvegetated during part of the year and dramatically changes the function of plants and soil microbes in the tilled areas. Repeated tillage, fertilization, and harvest permanently alter soil

character, resulting in reduced infiltration and increased surface runoff. These changes alter seasonal streamflow patterns by increasing high flows, lowering water tables, and reducing summer base flows in streams. Channelizing to reduce local flooding and alter the geometry of cropped lands also facilitates more rapid routing of water to the stream channel, thereby increasing peak flows downstream. Sediment yield from agricultural lands is typically greater than from prairie, forest, or wetland areas, and can reduce the quality of spawning gravels and the abundance of food organisms. Where riparian shading is lost or summer base flows are reduced, stream temperatures are increased. Nutrients, insecticides, and herbicides are typically elevated in streams draining agricultural areas, reducing water quality. Channelization, snag removal, revetments, and removal of riparian vegetation reduce habitat complexity, decrease channel stability, and alter the food base of the stream. As a result, incised and channelized streams in agricultural areas typically support smaller fish and fewer fish species.

1.5.4 Urbanization

Urbanization has affected only 2% of the land area of the Pacific Northwest, but the consequences to aquatic ecosystems are severe and long-lasting. The land surface, soil, vegetation, and hydrology are all significantly altered in urban areas. **As** development proceeds, the percentage of land covered by impervious surfaces increases, reducing the area available for infiltration and increasing surface runoff. Buildings, parking lots, roads, gutters, storm drains, and drainage ditches in combination quickly shunt precipitation to receiving streams, resulting in an increased magnitude and frequency of peak discharge and reduced summer base flow. Sediment delivery typically increases during construction activities. The total vegetated area is greatly reduced, and replacement vegetation, typically lawns and ornamental plants, require water, fertilizers, and pesticides. Riparian corridors frequently are constricted, disabling or altering riparian function. Loss of riparian vegetation and reduced base flows allow greater heating of streams during summer. In addition, the lack of recruitment of large wood combined with increased erosive potential of peak flows increase scouring of the streambed and downstream transport of wood, resulting in simplified stream channels and greater instability, These effects are exacerbated when streams are channelized and where banks are reinforced with concrete, rip-rap, or other hard structures. Water quality is adversely affected by inputs of fertilizer and pesticides washed from lawns and yards, discharge from sewage treatment facilities and industrial sources, and contaminated runoff from surface streets. The highly altered streams found in most urban areas provide poor habitat for fish and other aquatic biota.

1.5.5 Mining

Sand and gravel mining in streams and on adjacent floodplains have substantial effects on stream channels and hydraulic characteristics. In addition to the immediate morphological changes in stream channels caused by excavation, channels continue to exhibit instability, accelerated erosion, and altered substrate composition and structure after extraction has ceased. Downcutting of stream channels frequently follows gravel mining, often exceeding 4–6 meters in depth over periods ranging from months to decades. The downcutting and simplification of stream channels results in increased flood peaks, increased sediment transport, increased temperatures, and decreased base flows. The most direct impacts to salmonids are degradation and simplification of spawning and rearing habitats and increased turbidity. In addition, increased turbidity and decreased substrate stability influence lower trophic levels, upon which salmonids depend for food.

Mineral mining also substantially affects aquatic ecosystems. Although hydraulic mining is uncommon today, previously degraded habitats have not yet recovered and still exhibit excessive sediment transport, downcutting, and instability. For example, hydraulic mining (e.g., gold) from stream deposits and hillslopes dramatically altered stream channels, riparian zones, and floodplains. Recovery may take generations where channels have been modified and acid drainage, radioactive materials, and metals from mining wastes contaminate streams, Increased sediments, acidification, and chronic pollution from mine wastes seriously degrade aquatic habitats throughout the West. Streams receiving chronic metal pollution typically support few or no fish and degrade invertebrate assemblages.

1.5.6 Dams and Irrigation

Hydroelectric dams, impoundments, and withdrawing water for irrigation have significantly contributed to the decline of salmonids in the Pacific Northwest. Dams have impeded or blocked passage by adult and juvenile salmonids, and have caused gross changes in habitat conditions of rivers and streams. In the Columbia River basin, an estimated 55% of the total area and 33% of the total stream miles are no longer accessible to anadromous salmonids because of dams. At dams, injury and mortality to juveniles occurs as a result of passage through turbines, sluiceways, juvenile bypass systems, and adult fish ladders. Dams and reservoirs increase the time it takes juveniles to migrate to the

ocean, which increases exposure to predation. Attempts to bypass dams by barging and trucking may facilitate transmission of parasites and disease. Below hydroelectric facilities, nitrogen supersaturation may also negatively affect migrating salmon.

Hydrologic effects of dams and withdrawals for irrigation include water-level fluctuations, altered seasonal and daily flow regimes, reduced water velocities, and reduced discharge volume. Drawdowns and diversions reduce available habitat area and concentrate organisms, potentially increasing predation and transmission of disease. Dams have eliminated many spawning areas on large river systems and have created slackwater environments that are favorable to salmonid predators, including squawfish and a host of non-native piscivores. Impoundments alter natural sediment transport processes, causing deposition of fine sediments in slackwater areas, reducing flushing of sediments through moderation of extreme flows, and decreasing recruitment of coarse material (including spawning gravels) downstream of the obstruction. Return-flows from irrigated lands tend to have high sediment content, turbidity, and pesticide and fertilizer concentrations. Impoundments and water withdrawals also change the thermal regimes of streams. Temperatures may increase in shallow reservoirs and where return-flows from irrigation have been heated. Below deeper reservoirs that thermally stratify, summer temperatures may be reduced through release of hypolimnetic waters, but fall temperatures tend to increase as heated water stored during the summer is released. These changes in water temperatures affect development and smoltification of salmonids as well as influence the success of predators and competitors and the virulence of disease organisms. Dissolved oxygen concentrations may be reduced during both summer and winter from withdrawals for irrigation. In summer, high temperatures of return-flows reduce the oxygen-holding capacity of water; in winter, drawdown of impoundments may facilitate freezing, which diminishes light penetration and photosynthesis, potentially causing fish kills through anoxia.

1.5.7 Salmonid Harvest

Although this document focuses on the effects of human activities on salmonid habitats, it is important to acknowledge the effects fisheries have had on salmonid populations in the Pacific Northwest. Commercial, recreational, and tribal harvest of salmonids by humans constitutes a significant source of mortality for both anadromous and resident species, with harvest rates of adults in many fisheries exceeding 50% to 80% or more. Adverse effects of harvest on salmonids are particularly difficult to control in mixed-stock fisheries, where multiple species, stocks, and age classes are harvested together. Strong and weak stocks are harvested at comparable rates, as are fish of wild and hatchery origin. Mixed-stock fisheries are especially detrimental to naturally small populations or populations that have been depressed by human activities.

In addition to reducing total escapement of adult salmonids, harvest alters the age- and size-structure of salmonid populations. For many populations of anadromous salmonids, particularly species that spend several years at sea, mean size and age of harvested adults have steadily declined. This occurs because immature individuals are vulnerable to troll fisheries over a number of years. Consequently, larger and older individuals are harvested at a higher rate than individuals that mature earlier and at smaller size. Changes in size structure may also result from size-selective fishing gear. Changes in average size and age of individuals influences success of salmonid populations in several ways. Large size may confer several advantages including the ability to negotiate large barriers, higher fecundity, deeper deposition of eggs (and thus reduced risk of scouring during freshets), and utilization of larger, better oxygenated spawning gravels.

Harvest of salmonids can also influence the timing of certain life history events, including adult migrations, spawning, and juvenile migrations. Selective removal of early or late migrants can result in shifts in the timing of peak migration and spawning of a population. Finally, harvest of salmonids by humans can alter the fundamental structure of stream ecosystems through reduction of nutrient inputs from salmon carcasses as populations decline and average size of fish decreases.

1.5.8 Introduced Fish and Hatcheries

Introductions of non-native fish species and artificially propagated native salmonids pose additional risks to wild salmonids. Effects of species introductions on native fishes may include elimination, reduced growth and survival, and changes in community structure. Six mechanisms allow introduced fish to dominate or displace native fish: competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization. Introduced species may thrive best where extensive environmental modification has already occurred.

Artificial propagation of native salmonids has been used for decades to mitigate effects of habitat loss and to increase returns for harvest. Although artificial propagation may in some instances increase salmon and trout available for harvest, hatchery

introductions can result in a number of unintended and undesirable consequences for wild salmon and trout populations, for example, competition between hatchery and wild fish for food, habitat, or mates. Once in the ocean, large numbers of hatchery smolts may result in density-dependent decreases in survival and growth, although evidence of density-dependent effects in ocean environments is mixed. Other adverse effects of hatchery introductions include transmission of diseases between hatchery and wild populations, alterations of fish behavior (e.g., stimulation of premature smolt migration), and increased predation on wild fish (direct predation of hatchery fish on wild fish or attraction of predators). In addition to ecological effects, introduction of hatchery fish may lead to genetic changes in wild populations, including elimination of unique genomes in local stocks, loss of genetic variability between populations, and depressed fitness where introgression occurs.

The operation of hatchery facilities may adversely affect wild salmonid populations by contributing effluent with high concentrations of nutrients or disinfectant chemicals and by introducing pathogens. Hatchery weirs or diversion structures can impede the migration of wild stocks, and diversions of water for hatchery use reduces the amount available for wild stocks. Removal of wild fish for brood stock may threaten the genetic integrity of wild stocks, particularly for small or depleted stocks. Lastly, the removal of fish for brood stock decreases the amount of nutrients available in upstream reaches because salmon carcasses are not deposited.

Hatchery supplementation has social repercussions that influence wild salmonids directly and that affect the ability of managers to restore salmonid populations. Hatchery supplementation increases harvest pressure on wild populations in mixed-stock and terminal fisheries, particularly during years when survival of hatchery fish is low due to poor environmental conditions, and fisheries become overcapitalized. In addition, once commercial and sport fishers have invested large sums of money in fishing gear, they may resist increased fishing restrictions, making it difficult for managers to enact stricter protection for wild stocks. Finally, the long history of hatchery programs in the United States has instilled a perception in the public that habitat losses or degradation can be mitigated through artificial propagation, a perception that may impede implementation of more ecologically sound restorative activities.

1.6 Effects of Atmospheric and Ocean Circulation

Marine productivity depends on atmospheric and oceanic circulation and strongly affects abundance of salmonids and other fishes. Surface currents of the northeastern Pacific are dominated by the "West Wind Drift," which flows west-to-east across the Pacific and bifurcates as it approaches North America into the Alaska Current flowing north and the California Current flowing south. Changes in climatic conditions affect the behavior of the West Wind Drift. In years where a strong Aleutian Low Pressure system develops off the south coast of Alaska, typical of El Niiio conditions, a greater percentage of cold, nutrient-rich water is diverted north into the Alaska Current. When the Aleutian Low is weaker, typical of La Niiia years, more water from the West Wind Drift is diverted south towards California. These shifts, combined with changes in prevailing wind directions and upwelling patterns, can substantially affect conditions for salmonids entering the ocean. Changes in surface currents and upwelling strength influence temperature, salinity, and nutrients, thereby affecting the abundance of food available to juvenile salmonids, the number and distribution of predators and competitors, and the transport of smolts entering the ocean (along-shore versus off-shore). Recent evidence suggests that when ocean conditions are poor for salmonids in the Pacific Northwest, conditions are favorable to Alaskan stocks and vice versa.

Cycles in marine productivity can mask the effects of habitat degradation in freshwater environments or other stressors of salmonid populations. Long-term trends in the ability of freshwater environments to support salmonids may not be evident during periods of favorable oceanic conditions, particularly for populations augmented by hatchery fish. However, as ocean conditions shift towards less favorable conditions (particularly for hatchery fish), increasing pressure from overcapitalized fisheries can dramatically reduce the abundance of wild stocks.

1.7 Practices For Restoring and Protecting Salmonids and Their Habitats

Virtually all land-use and water-use practices have some effect on aquatic ecosystems, as do the harvest of salmonids and the introduction of non-native and hatchery fish. However, there are numerous opportunities, through planning and specific practices, for minimizing these effects or mitigating for past damage. Regardless of the activity, emphasis should be placed on preventing (rather than mitigating) damage, particularly in those areas where high-quality habitats and stable salmonid populations remain.

Impacts of harvest on wild salmonids can be best controlled through terminal and bay fisheries that target adults as they return to their natal streams.

Terminal fisheries provide greater protection for weak stocks by targeting hatchery runs instead of wild stocks, by allowing late-maturing fish to reach maturity, and by reducing the incidental mortality of subadults. These targeted fisheries avoid problems associated with managing for indicator or weak stocks traditionally used in open ocean, mixed-stock fisheries. Harvest methods can also be changed to target hatchery stocks and reduce incidental mortality of wild populations. Traps, fish wheels, and hook-and-line angling all cause lower mortality than gillnets or trolling. Special sport angling restrictions, including catch-and-release angling, minimum size or slot limits, and bag limits may further reduce mortality or minimize size-selective harvest. Accurate monitoring of escapement levels of specific stocks is essential for establishing exploitation levels that ensure the long-term persistence of individual stocks.

Growing evidence of the adverse ecological, genetic, and social consequences of hatchery operations suggests substantial modification, curtailment, or elimination of hatchery programs for salmonids would benefit wild populations, though not without adverse short-term social and economic impacts. Emphasis of hatchery programs is beginning to shift from increasing fish harvest to conserving endangered species or supplementing weak stocks, though the risks of using hatcheries for these purposes are still being debated. Potentially beneficial hatchery programs include those to re-establish native species into waters where fish have been extirpated by human activities; those to sustain a presently overharvested fishery through a planned program of downsizing and transition to other employment or from reliance on hatchery fish to reliance on wild fish; and those to augment weak stocks (put-and-grow stocking) in waters having little or no reproductive habitat but substantial productive potential where stocking will not harm indigenous biota, Impacts of introduced (non-native) fish species on wild salmonids can be minimized by ceasing the stocking of non-native fish into waters that contain wild salmonids, by direct removal by piscicides and electrofishing, and by indirect removal through use of unrestricted catch limits. Restoring streams and rivers to their natural temperature and flow regimes may reduce the spread of non-native species into salmonid streams.

A number of large-scale habitat restoration programs are currently underway or in the planning stages. In the Kissimmee River, Florida, steps are being taken to re-establish natural channel configuration, floodplains, and hydrologic regimes. In the Elwha (Washington) and Rogue River (Oregon) basins, dam removal has been proposed to restore salmon habitats and remove barriers to migration. Elsewhere, the impacts of dams are being reduced by assuring instream flows, especially at critical times; screening turbine intakes; and improving bypass systems. Direct impacts to river channels can also be minimized by retaining large woody debris and channel complexity and by restricting snagging and channelization.

Impacts of forest practices can be reduced through longer rotations; selective harvesting instead of clear-cutting; logging during the dry season or when the ground is frozen; use of high lead, skyline, and helicopter logging instead of ground-based equipment; use of designated skid trails; minimizing site-preparation practices that compact or scarify soils; retention of riparian buffer zones along streams; designation of no-cut zones in areas prone to mass failures; careful placement and maintenance of roads; and decommissioning and reseeding of roads when logging is completed. These activities function to minimize the percentage of the watershed in a disturbed state, reduce the total area of ground disturbance and soil compaction, minimize surface runoff and sediment loads, and protect and preserve the function of riparian zones.

The effects of range practices can be reduced by resting pastures, decreasing numbers of livestock, controlling livestock distribution through fencing of riparian zones or watering of stock away from riparian areas, controlling forage use, controlling season of use, and determining the kind of livestock best suited for the area. These practices can serve to reduce grazing stress, ensure that sufficient vegetative cover remains after the grazing season, promote the re-establishment of riparian vegetation (particularly woody shrubs and trees), and keep stock out of riparian zones, although site-specific conditions will determine their relative effectiveness.

Agricultural practices and policies that promote water and soil conservation and that reduce chemical application can all reduce effects on aquatic ecosystems. Examples include switching to crops that do not require irrigation, ditch lining and drip irrigation, screening of intakes for irrigation, increasing vegetative cover (e.g., permanent rather than annual crops), conservation tillage, planting grass in water ways (for soil conservation), organic farming, integrated pest management, and increasing tax relief for farmers employing conservation practices as well as penalties for those who do not.

Most of the impacts of gravel mining relate to changes in channel morphology that create channel instability, cause bedload movement, and increase sedimentation. Consequently, these effects can be most productively reduced by eliminating instream mining. Bar scalping instead of below-surface extraction has been used to minimize turbidity and direct damage to spawning habitats; however, changes in channel morphology are likely to occur as

water levels rise. Abandoned gravel mines in floodplain areas may offer opportunities for increasing off-channel habitats. Effects of mineral mining can be reduced by burying toxic materials below the root zone, by rehabilitating the site using created natural contouring and re-established natural vegetation, and by controlling mining-generated solids and liquids with containment structures and waste treatment.

Urbanization permanently alters many natural watershed processes, and in some cases, little may be done to mitigate effects. Thus, the most effective way to minimize impacts is through careful land-use planning that minimizes the total impervious area and that precludes development along streams and in natural floodplains. Sewage treatment and programs to foster water conservation, minimize chemical applications, and prevent toxic materials from being dumped into drainage structures can reduce impacts of urbanization to water quality.

Finally, because the condition of aquatic habitats is ultimately tied to resource consumption—the use of water, electricity, wood products, meat and wool, food and nonfood crops, and mineral resources—conservation of salmonids will require re-examination of fundamental aspects of our culture, including actions of individuals, population and economic policies, and ethical concerns. Policies that promote conservation need to be encouraged while those that foster waste and overconsumption need to be discouraged. Education is central to increasing the awareness of citizens as to how their actions directly or indirectly affect salmonids and their habitats.

1.8 Relevant Federal Laws for Protecting and Restoring Salmonid Ecosystems

Several federal laws, notably the Clean Water Act (CWA), the Endangered Species Act (ESA), the National Environmental Policy Act (NEPA), and the Food Security Act (FSA), are or could be employed to protect aquatic and riparian habitats on nonfederal lands. The goals of the CWA are to restore and maintain the chemical, physical, and biological integrity of the Nation's waters; to eliminate discharge of pollutants into waters; to attain water quality that provides for the protection and propagation of fish, shellfish, and wildlife; and to develop and implement area-wide waste treatment management to control pollutant sources. The NEPA declares a national policy that encourages harmony between humans and their environment, reduces environmental damage, and improves understanding of ecological systems. The **ESA** seeks to conserve the ecosystems upon which threatened and endangered species depend and to provide a program to conserve listed species and their ecosystems. The FSA encourages conservation by making ineligible for Federal price supports, loans, crop insurance, or disaster payments any landowner who produces a crop on highly erodible lands or on converted wetlands. Each of these laws may be used to provide Federal leadership in furthering the goals of habitat conservation.

19 Monitoring Conservation Efforts

The success of salmonid conservation efforts depends on a rigorous monitoring program for determining whether conservation plans are being implemented and if they are effective. Examples drawn from existing programs to monitor wetland permits, forest plans, point-source discharges, and rural best-management plans indicate a number of common shortcomings. These include inadequate funds and staff, unclear objectives and criteria, failure to use remote sensing and site visits, and lack of computerized data systems. Periodic status reports and peer reviews are essential to successful monitoring as well. To make a monitoring program most useful and cost-effective, it must be regional, have a statistical design, and be based on quantitative physical, chemical, and biological indicators. Consistency of indicators between Federal and State monitoring programs is also essential.

2 Introduction

The structure of aquatic ecosystems—the physical habitats, the material and energy resources, and the associated biological communities—arises from complex interactions among numerous processes that occur in upland areas, within riparian zones, and in stream channels, lakes, or estuaries. Physical processes act in concert with vegetative characteristics to provide the physical and chemical context within which aquatic systems develop and to regulate the exchange of material and energy from the watershed to the stream channel. Biological processes both influence the conversion of material and energy as well as govern the relationship of organisms to one another and to their environment. Collectively, physical, chemical, and biological processes give rise to ecosystem structures, which in turn exert influence back on those processes. These ecosystem-shaping processes operate over a wide range of temporal and spatial scales. Protection and recovery of salmonid habitats fundamentally depends on maintaining and restoring, in both space and time, the natural rate or frequency of occurrence of these processes and the ecosystem structures to which they give rise.

Part I of this document comprises a technical foundation for understanding salmonid conservation principles and developing salmonid conservation plans in an ecosystem context. We intentionally focus on freshwater habitats but recognize that many other natural and anthropogenic factors, which we only discuss briefly, influence greatly the abundance of salmonids, including fish harvest, hatchery practices, habitat conditions in near-shore areas, and natural variation in ocean productivity. Conclusions are based on our assessment of the scientific literature. Because some topics are thoroughly discussed in this literature and others are not, certain sections of the document are relatively complete and robust, but others are more sparse.

2.1 Scope

Geographically, the scope of this document is limited to the Pacific Northwest region, including portions of California, Oregon, Washington, and Idaho that presently support or historically supported salmonid populations. Many general concepts and processes examined, however, are equally relevant outside this region. Discussion of specific habitat requirements is restricted to salmonid species that are endemic to the Pacific Northwest (Table 2-l), including the five Pacific salmon (chinook, coho, sockeye, chum, and pink salmon), trout and char with both resident and anadromous forms (rainbow and cutthroat trout, Dolly Varden char), and strictly resident species (bull trout, mountain whitefish).

In the remainder of Chapter 2, we discuss evidence of widespread declines in salmonid abundance that indicate region-wide degradation in habitat quality and ecosystem condition. We then identify strategies for restoring salmonid habitats.

Table 2-1. Common and scientific names of salmonids native to the Pacific Northwest.

Common name	Scientific name
Pink salmon	*Oncorhynchus gorbuscha* (Walbaum)
Chum salmon	*Oncorhynchus kefa* (Walbaum)
Coho salmon	*Oncorhynchus kisufch* (Walbaum)
Sockeye salmon	*Oncorhynchus nerka* (Walbaum)
Chinook salmon	*Oncorhynchus fshawyfscha* (Walbaum)
Mountain whitefish	*Prosopium williamsoni* (Girard)
Cutthroat trout	*Oncorhynchus clarki* (Richardson)
Rainbow and steelhead trout	*Oncorhynchus mykiss* (Walbaum)
Bull trout	*Salvelinus confluenfus* (Suckley)
Dolly Varden	*Salvelinus malma* (Walbaum)

These strategies emphasize the importance of maintaining natural watershed processes, providing for the diverse life-history requirements of salmonids, and re-establishing connectivity between salmonid habitats across the landscape.

In Chapters 3 and 4 we review physical, chemical, and biological processes that occur within watersheds, that influence the quality and quantity of available salmonid habitat, and that need to be maintained to ensure the persistence of salmonid stocks. Some physical and chemical processes (Chapter 3) shape stream habitats over long time periods (e.g., glaciation, volcanism) and others operate in relatively short time scales (e.g., floods, droughts, landslides). Biological processes (Chapter 4) encompass those occurring at the level of the individual organisms (e.g., physiology, behavior), populations (e.g., life history, adaptation), and communities (e.g., disease, predation, parasitism, competition). Chapters 3 and 4 provide sufficient detail about ecological processes that the effects of anthropogenic disturbances on salmonids and their habitat can be understood and evaluated. Chapter 5 describes habitat requirements specific to each stage of the salmonid life history and general characteristics of healthy aquatic and riparian systems, including physical habitat structure, streamflow, stream temperature, water quality, and important biological elements.

In Chapter 6, we discuss the effects of human activities on watershed processes and the resulting impacts on salmonids and their habitats. The discussion focuses on effects of land-use practices including forestry, livestock grazing, agriculture, mining, and urbanization. Effects of water uses, including hydroelectric dams and irrigation impoundments, are also reviewed. Although the effects of other human activities such as fish harvest, hatchery supplementation, and introduction of non-native species are largely outside the scope of this project, these issues are discussed to provide an appropriate context from which to view habitat-related issues. Chapter 7 briefly reviews general circulation patterns and the dominant physical processes controlling conditions in the northeastern Pacific Ocean. This chapter also discusses how ocean conditions influence abundance and distribution of aquatic organisms, including anadromous salmonids, and the relevance of these natural production cycles to the conservation of freshwater habitats of salmonids.

Chapter 8 identifies management systems and practices that are designed to minimize effects of human activities on salmonid habitats, with emphasis on forestry, range, and agricultural practices, as well as urban planning. Chapter 9 summarizes four Federal laws and associated amendments that pertain to conserving and protecting aquatic species and their habitats on nonfederal lands. These include the Endangered Species Act (ESA), the Clean Water Act (CWA), the National Environmental Policy Act (NEPA) and the Food Security Act (FSA).

The importance of a rigorous implementation and monitoring program for aquatic resource conservation is discussed in Chapter 10, wherein limits and inadequacies of previous programs are used as examples. The value of monitoring several physical, chemical, and biological indicators is also discussed.

2.2 Historical Background and Evidence of Habitat Degradation

Many Pacific salmon stocks have been depleted to the point that continued declines will likely result in additional listings under the Endangered Species Act (ESA) or local extirpations. Although ample evidence documents historical declines in Pacific salmonids (Ebel et al. 1989), the landmark paper by Nehlsen et al. (1991) alerted both scientists and the public to the extent of these declines. Summarizing the status of Pacific salmon of Washington, Oregon, Idaho, and California, Nehlsen et al. (1991) listed 106 stocks (unique populations) that have been extirpated from the region and 214 stocks that are at high or moderate risk of extinction or of special concern. Huntington et al. (1996) concluded that only 99 stocks of native anadromous salmonids in the region have populations greater than one-third their historical abundance, and just 20 stocks are at levels greater than two-thirds of their former abundance.

Since 1985, tribes, professional fishery societies, and conservation organizations have petitioned the National Marine Fisheries Service (NMFS) to list 24 stocks as threatened or endangered. To date four of these have been listed (Sacramento River winter chinook, Snake River sockeye, and Snake River spring/summer and fall chinook salmon). In addition, NMFS has recommended listing coho salmon as threatened throughout all of California and most of Oregon (NMFS 1995a). NMFS has added coho stocks in southwestern Washington and Puget Sound to the candidate species list; information is currently insufficient to warrant listing, but specific risk factors have been identified, and concerns need to be resolved before a final status determination is made. Commercial and sport ocean-harvests in the Pacific Northwest have been sharply curtailed in recent years because of dwindling numbers of salmon and concern for wild salmon stocks. In 1994, the ocean commercial and recreational fisheries for coho and chinook salmon were completely closed from Washington to Cape Falcon, Oregon, with the exception of a limited treaty Native American troll fishery for chinook salmon off northern Washington.

South of Cape Falcon to central California, fishing for coho salmon was restricted to recreational catch, and no retention of coho salmon was allowed after May 1 (PFMC 1995). In 1995, the commercial and recreational coho salmon fishery was completely closed, as was the chinook fishery from northern Washington to Cape Falcon. The fishing seasons for chinook salmon were severely restricted from Humbug Mountain to central California.

A number of factors have been implicated in the decline of Pacific salmonids including dams, overexploitation, disease, natural predation, artificial propagation, climatic variation, and the destruction and alteration of habitat. The relative importance of each of these factors in influencing salmonid populations varies across the region; however, habitat loss and modification are believed to be the major factors determining the current status of salmonid populations (FEMAT 1993). Nehlsen et al. (1991) concluded that present or future habitat degradation (including mainstem passage and flow problems) represents a significant threat to 90% of populations of anadromous Pacific salmonids identified as at high or moderate risk of extinction or of special concern. Similarly, Miller et al. (1989b) reported that physical habitat degradation was identified as a causal factor in 73% of fish species extinctions in North America during the past 100 years. A wide range of land- and water-use practices have contributed to the degradation of aquatic habitats, including timber harvesting, livestock grazing, agriculture, mining, urbanization, road construction, and construction of dams for hydroelectric power, irrigation, and flood control. Alterations in riverine systems that result from these activities include 1) changes in water quantity or flow because of water storage and irrigation or other withdrawals; 2) direct modification of channel morphology and riparian ecosystems by dams, reservoirs, channelization, draining and filling of wetlands, and dredging for navigation; 3) land-use practices that alter upland and riparian vegetation and, thus, the delivery or water, sediment, organic matter, and nutrients to streams; and 4) excessive point and nonpoint source pollution (Doppelt et al. 1993). Over time, land-use practices have substantially decreased the physical and biological complexity of ecosystems, thereby diminishing the ability of ecosystems to self repair when perturbed (FEMAT 1993).

Regional patterns in declines of salmonids and other fishes in the Pacific Northwest suggest that deterioration of freshwater habitats is widespread, with certain regions being particularly degraded. The 214 at-risk salmon stocks identified by Nehlsen et al. (1991) are distributed throughout Washington, Oregon, California, and Idaho. At least two-to-three species of fish (including nonsalmonids) are extinct or at risk of extinction in most areas of the Pacific Northwest, indicating that species losses are not isolated occurrences (Figure 2-1) (Frissell 1993b). Nawa (1994) examined population trends for 228 stocks of spring and fall chinook salmon over the period from 1940–1993 and found that 34% were extinct or nearly extinct, 24% were declining, and only 8% were not declining. Other chinook stocks were either hatchery-influenced or had unknown status. Bisson et al. (1992b) report that more Alaskan stocks of chinook, coho, chum, and steelhead increased than decreased from 1968 to 1984 (Figure 2-2). Conversely, in Washington, the Columbia River Basin, and coastal Oregon and California, declining stocks outnumbered increasing stocks for all four species examined, though the majority of stocks exhibited no significant trend over the 16-year period (Figure 2-2). Frissell (1993b) examined native fish taxa that are considered extinct, endangered, or threatened in drainage basins of the Pacific Northwest and reported a north-south gradient in the degree of endangerment (Figure 2-1); mean percentages of the native taxa considered to be extinct or at risk of extinction were 13.5% in Washington, 33.0% in Oregon, and 48.0% in California. This pattern is largely influenced by the basin-specific populations of seven widely distributed species of anadromous salmonids rather than locally endemic species such as suckers, pupfishes, and minnows. Another status review of the five Pacific salmon and the anadromous steelhead and cutthroat trout (TWS 1993) indicates a similar latitudinal gradient in the degree of endangerment for most of these species (Figure 2-3 through Figure 2-11). The general north-south gradient in salmonid declines likely reflects several factors. First, the environments in the southern portion of the salmonids' range are more extreme, with specific habitat attributes (e.g., temperature, streamflow) approaching the tolerable limits for the species. Second, there has generally been a higher degree of habitat modification in the southern part of the range. And finally, the influence of changing oceanic conditions varies with latitude (see Chapter 7).

In addition to the north-south gradient in species declines, several subregions and localized areas have an especially high degree of species endangerment. The risk of extinction is greatest in the upper Columbia—with multiple large hydropower dams and large-scale water diversions—as well as in many other undammed coastal and Puget Sound streams (Figure 2-12). Prior to development, 10–16 million salmon returned to the Columbia River to spawn each year; however, recent estimates suggest that fewer than 0.5 million wild fish now spawn in the Columbia River and its tributaries (NPPC 1992b). Coho salmon historically were abundant throughout

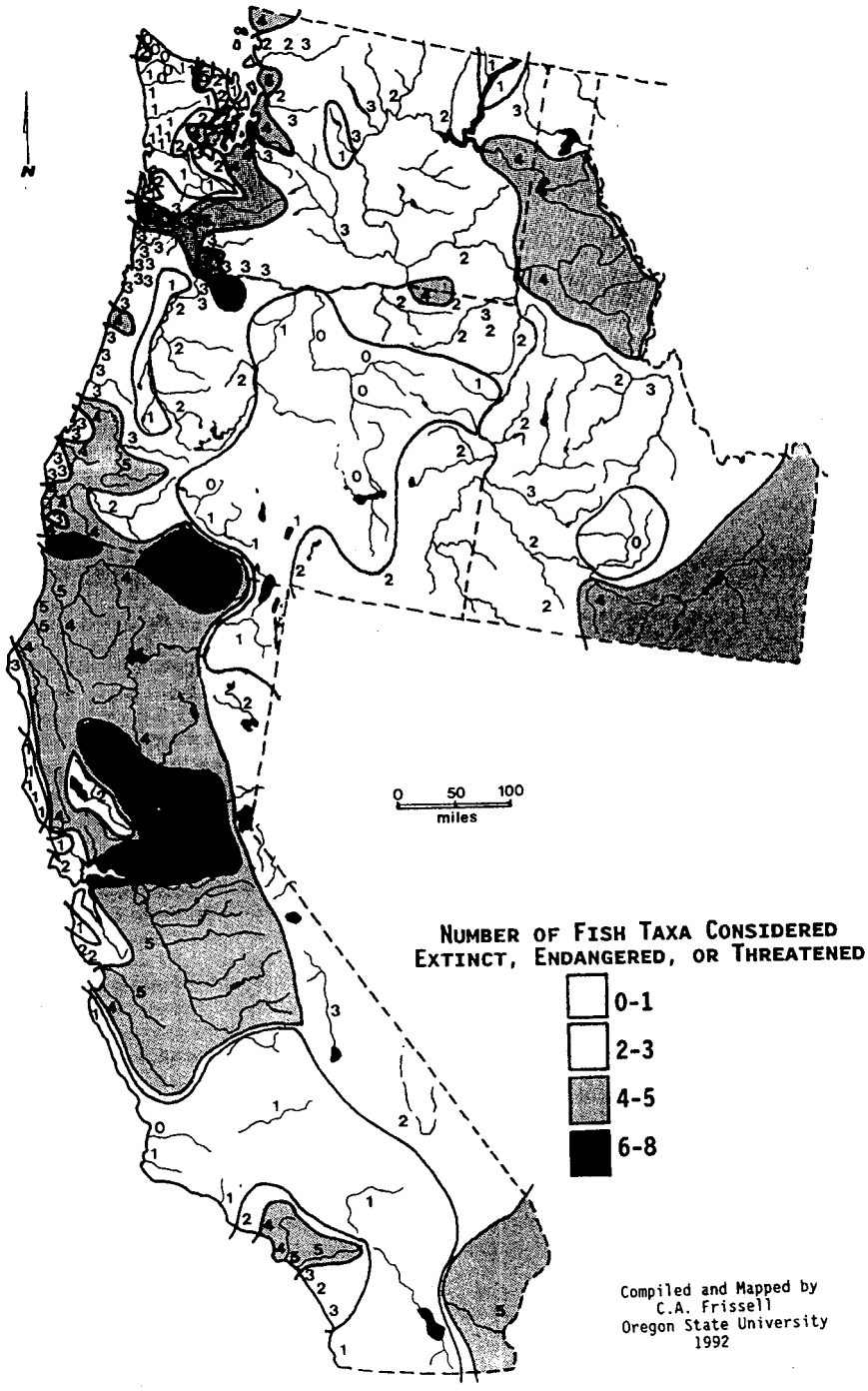
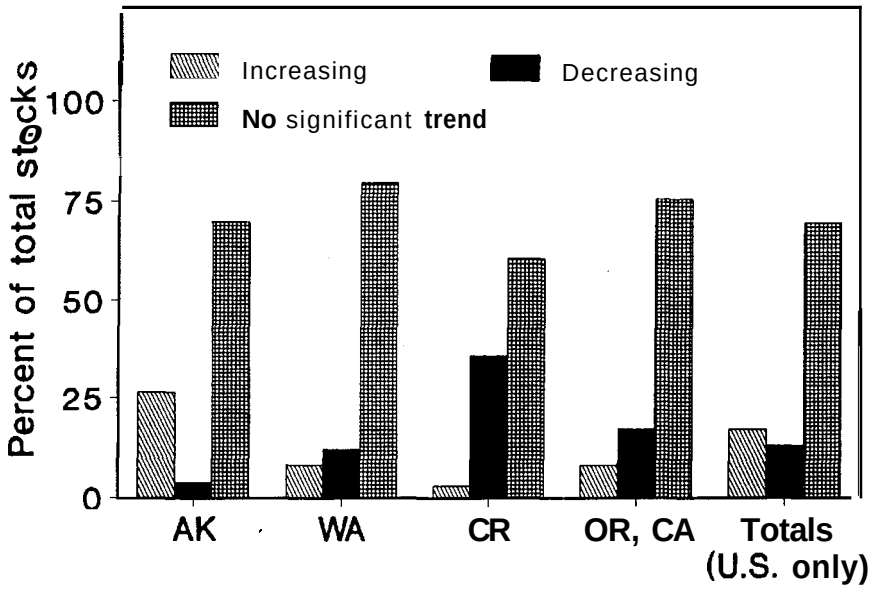


Figure 2-1. Number and location of fish species considered extinct, endangered, or threatened in the Pacific Northwest and California. From Frissell (1993b). Reprinted by permission of Blackwell Scientific Publications, Inc.



Region	Chinook UP	Chinook DN	Coho UP	Coho DN	Chum UP	Chum DN	Steelhead UP	Steelhead DN
Alaska	43%	1%	15%	11%	3%	13%	17%	0%
Coastal WA	12%	32%	9%	0%	6%	15%		
Columbia R. Basin	3%	39%	0%	45%	0%	33%	8%	25%
Coastal OR, CA	19%	12%	2%	17%	11%	11%	20%	40%
U.S. Total	20%	22%	6%	17%	4%	14%	11%	23%

Figure 2-2. Trends in the abundance of wild stocks of chinook salmon (Oncorfiynchus tshawytscha), coho salmon (*O.* kisutch), chum salmon (*O.* kefa), and steelhead (O. mykiss) from river systems along the Pacific coast. UP = percentage of stocks significantly increasing, DN = percentage of stocks significantly decreasing. From Bisson et al. (1992b) based on data from Konkel and McIntyre (1987). Reproduced with permission of the publisher.

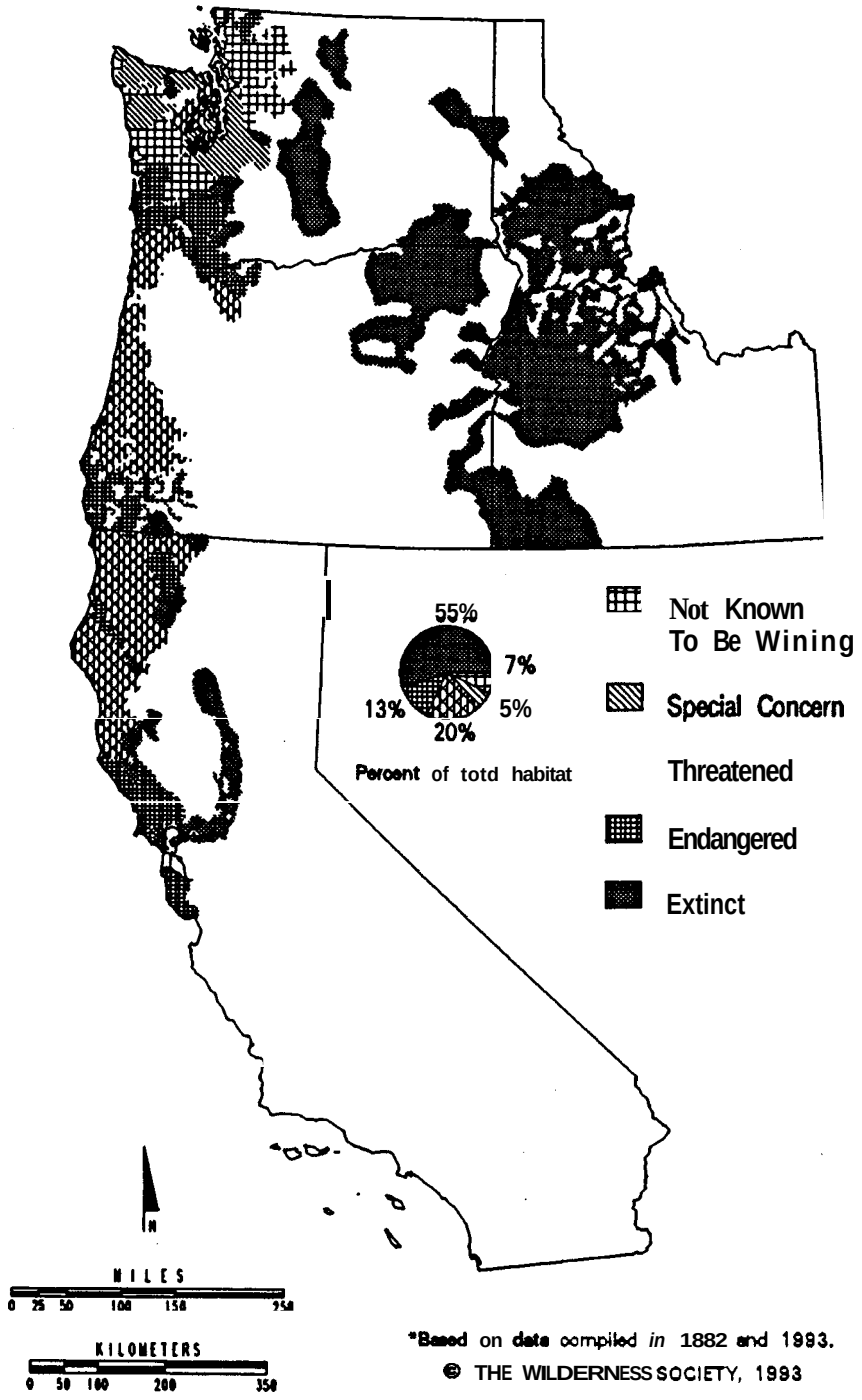


Figure 2-3. Status of coho salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

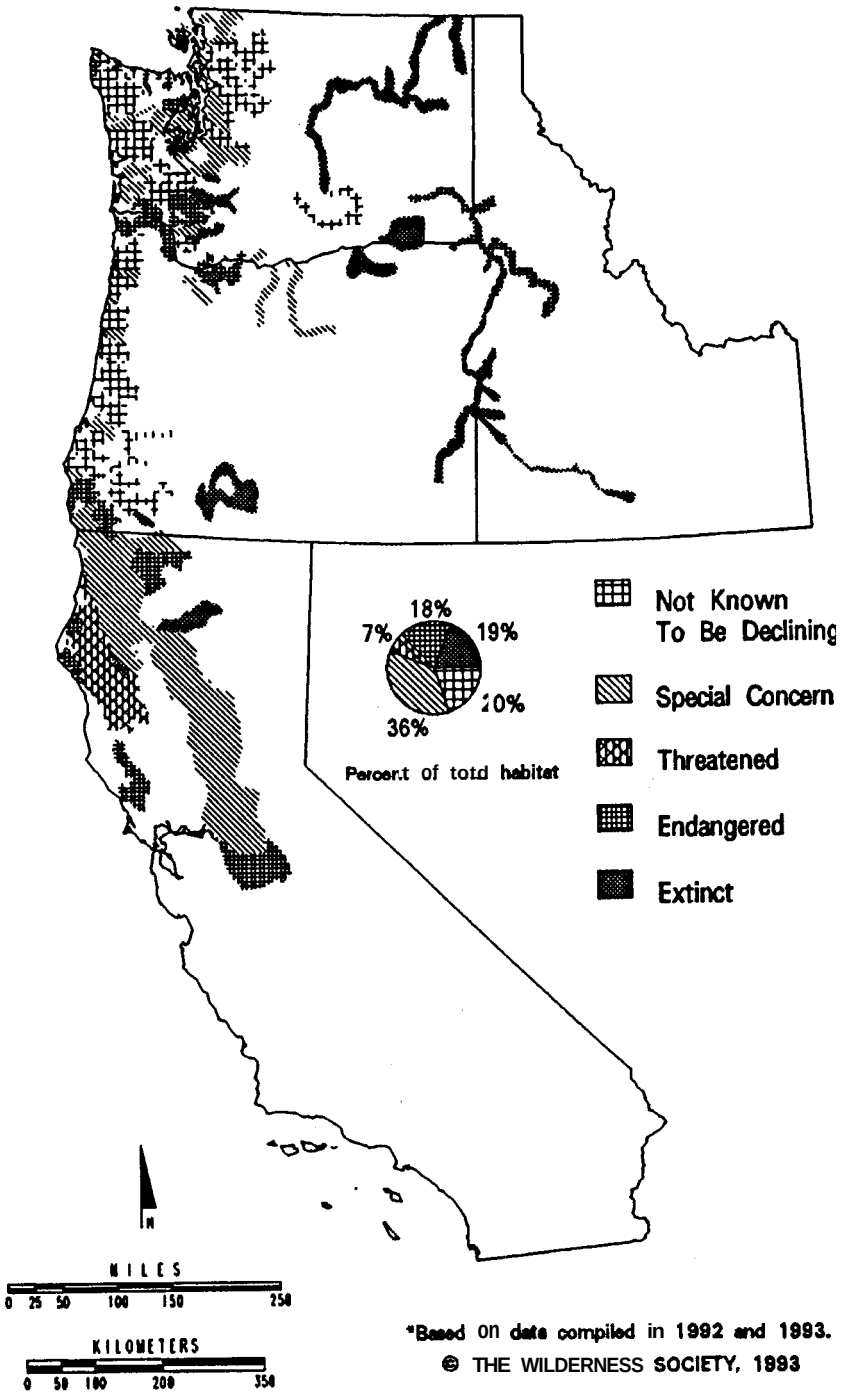


Figure 2-4. Status of fall chinook salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

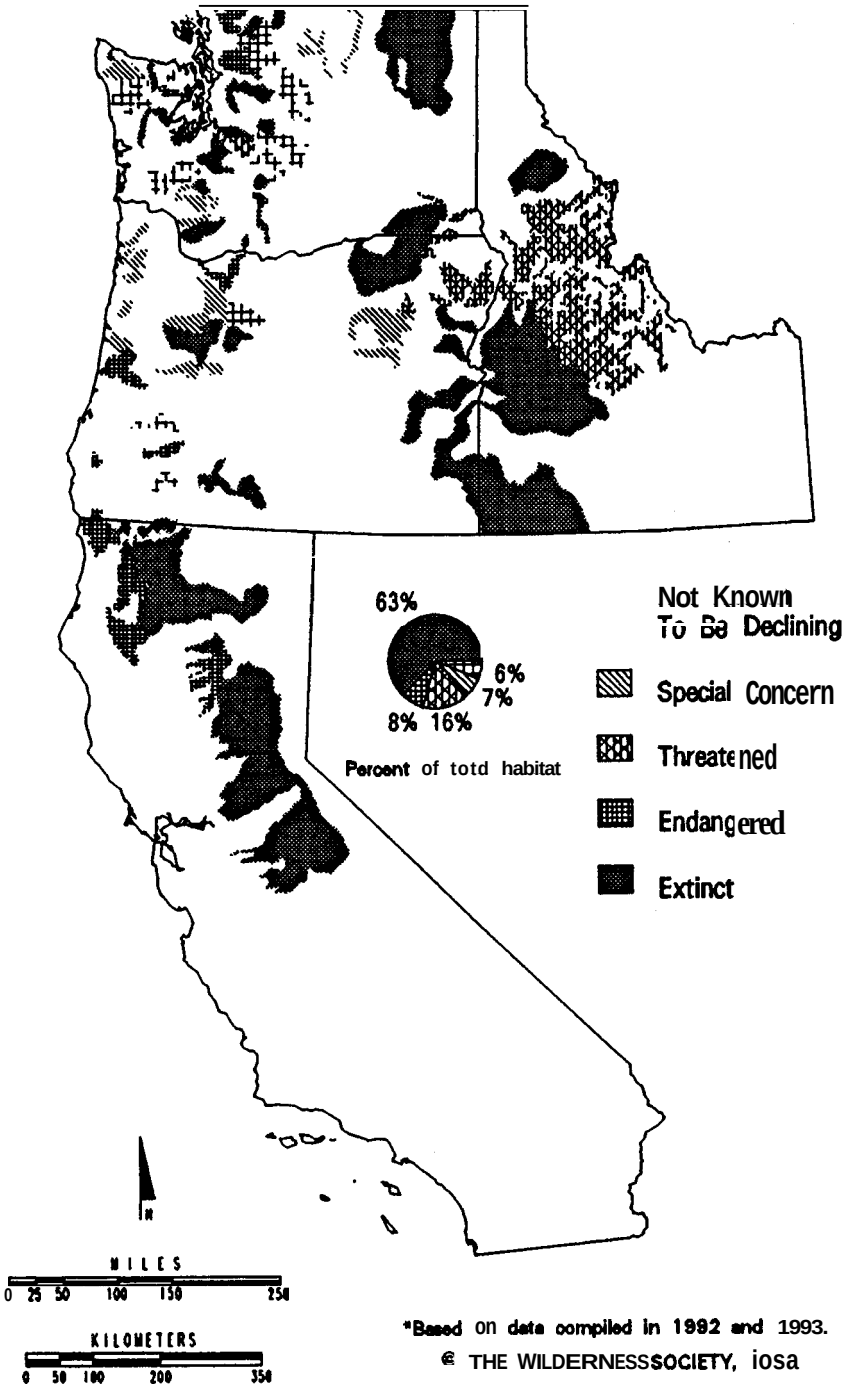


Figure 2-5. Status of spring and summer chinook salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

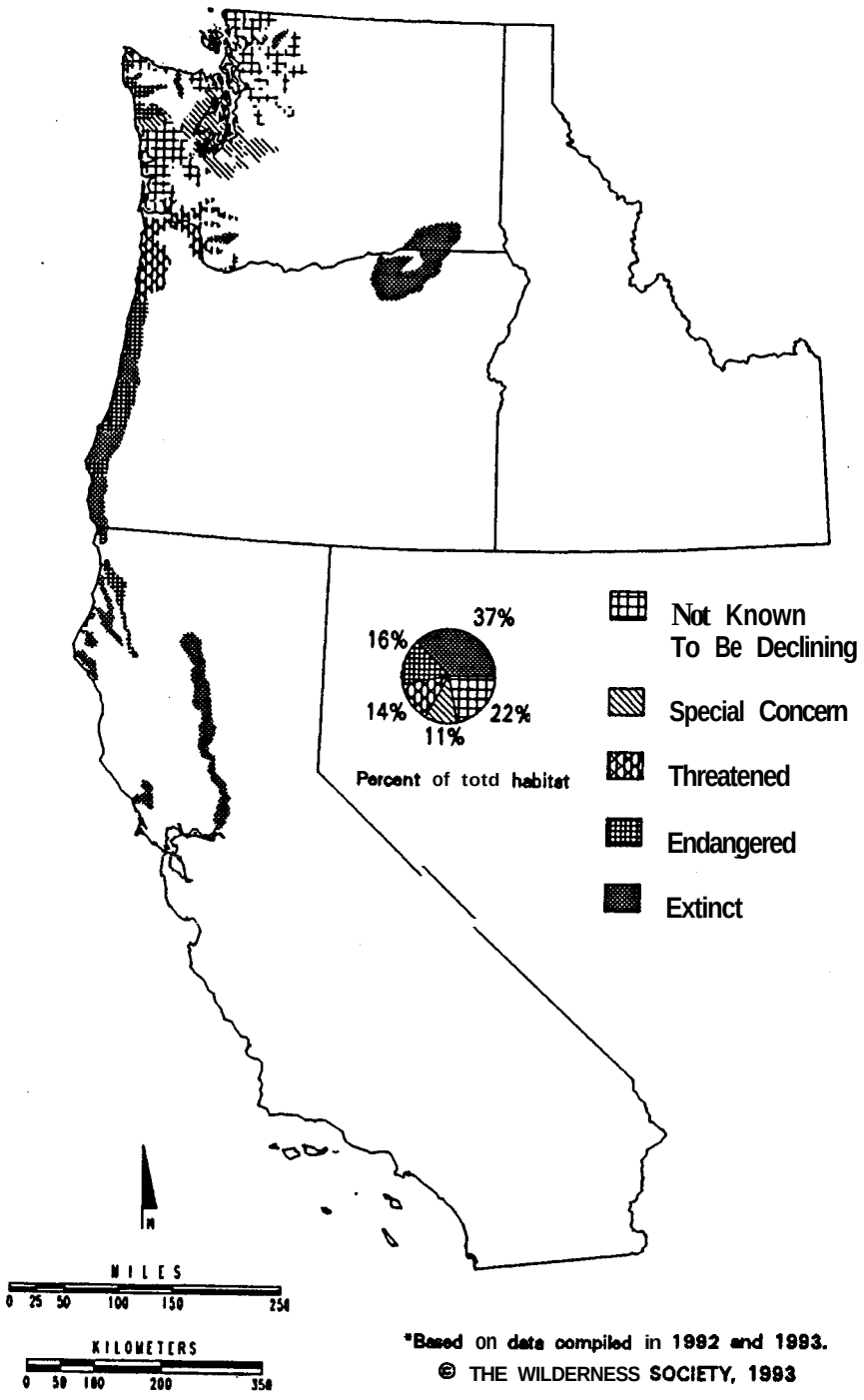


Figure 2-6. Status of chum salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

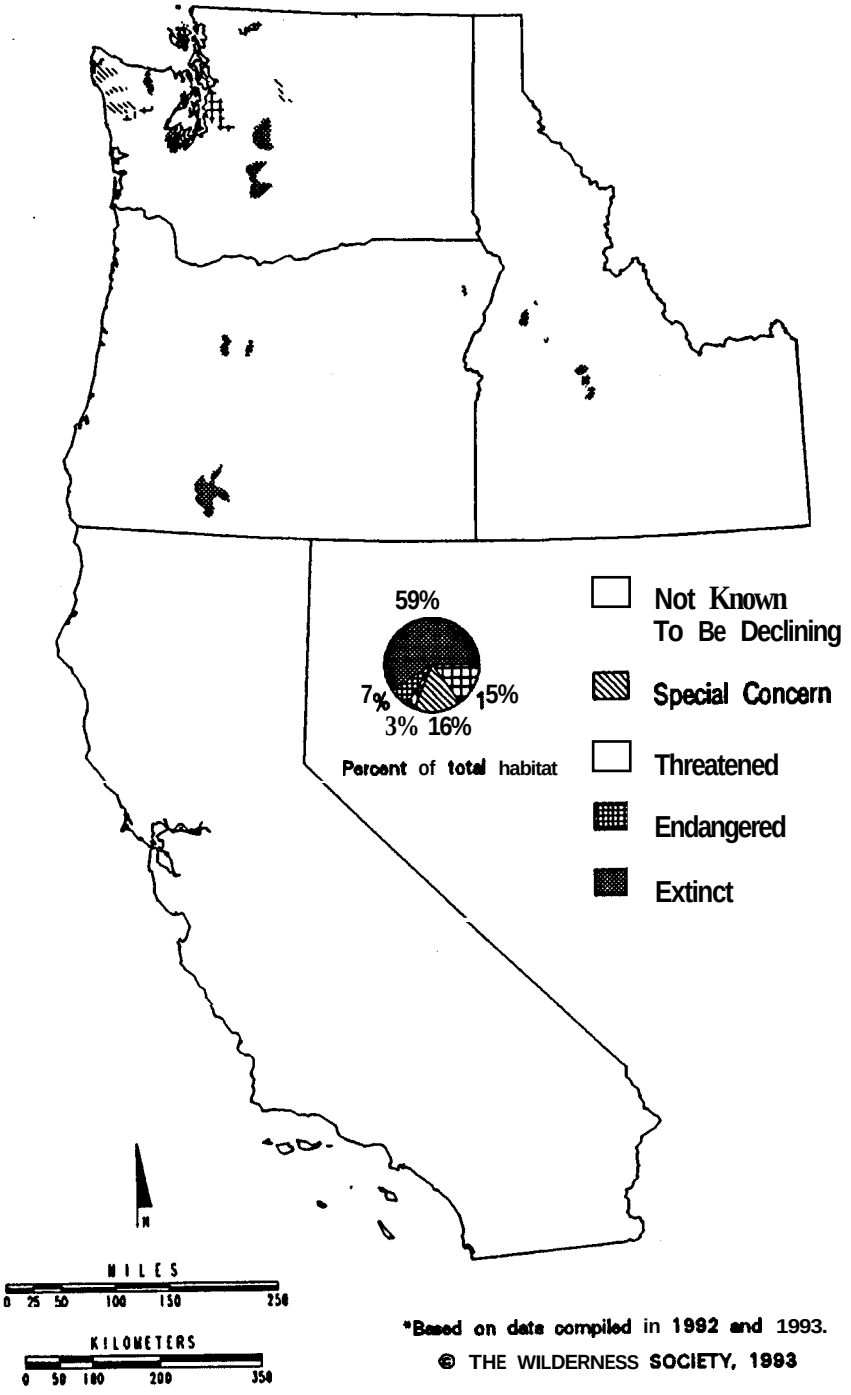


Figure 2-7. Status of sockeye salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

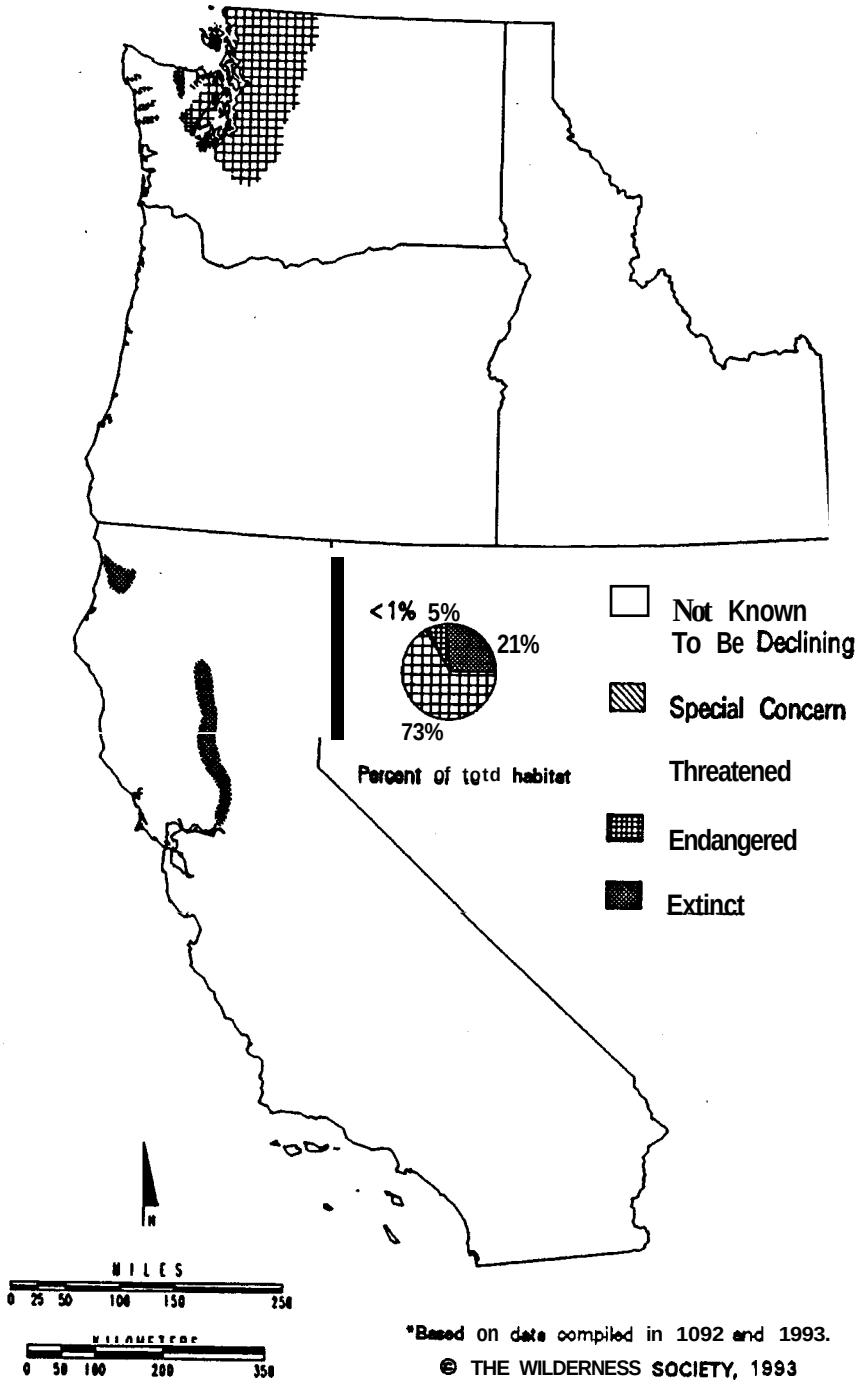


Figure **2-8.** Status of pink salmon in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

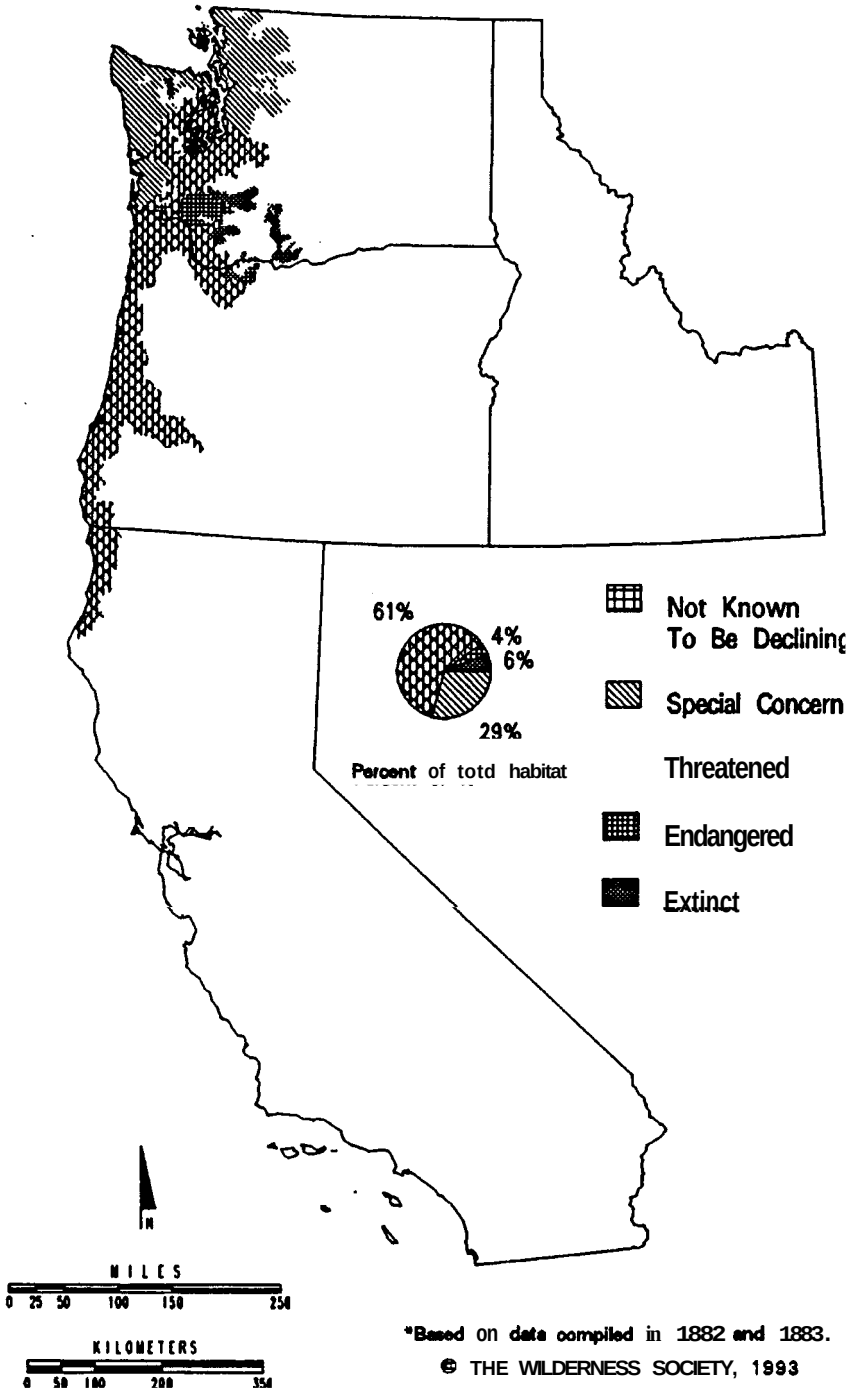


Figure 2-9. Status of sea-run cutthroat trout in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

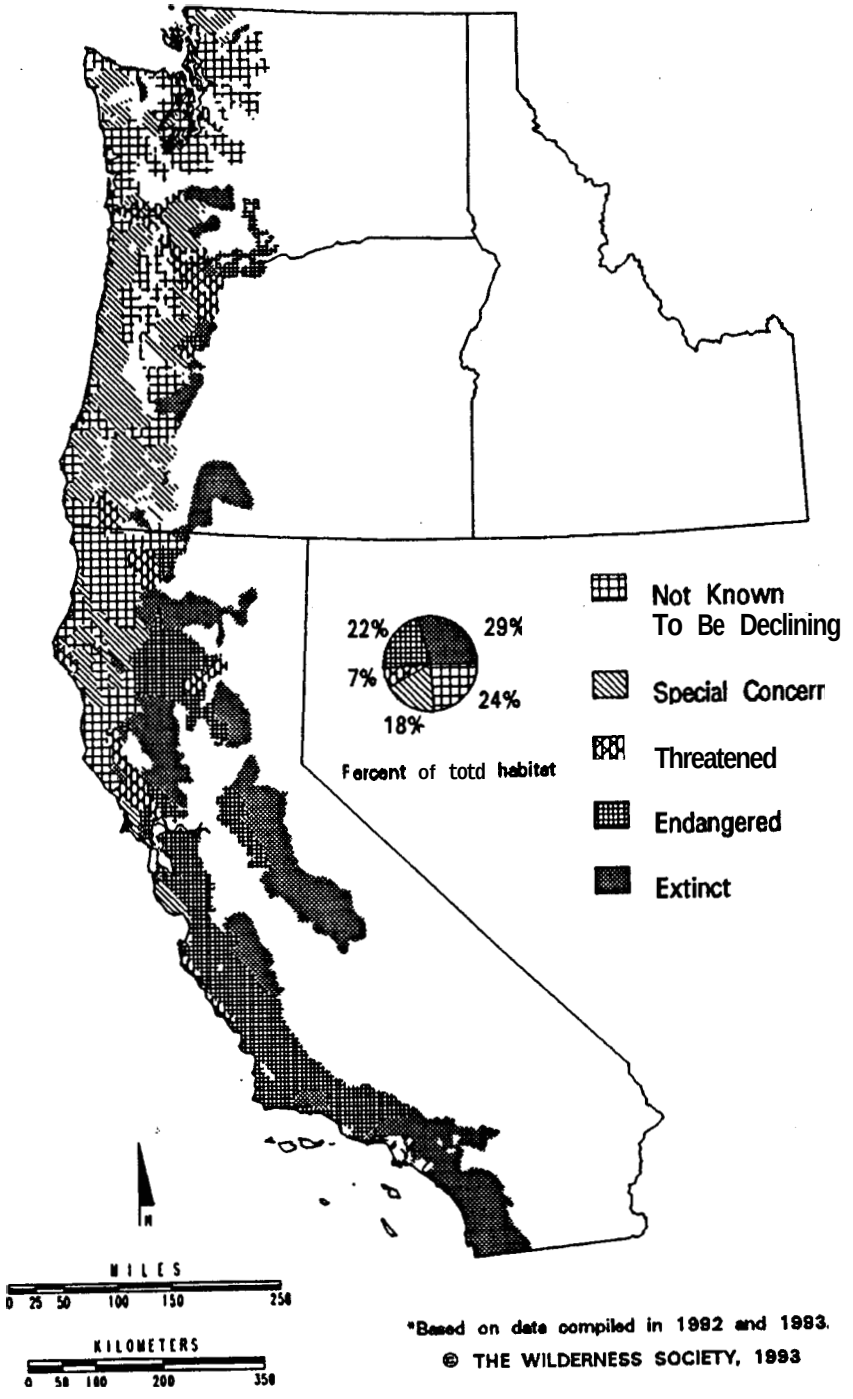


Figure 2-10. Status of winter steelhead in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

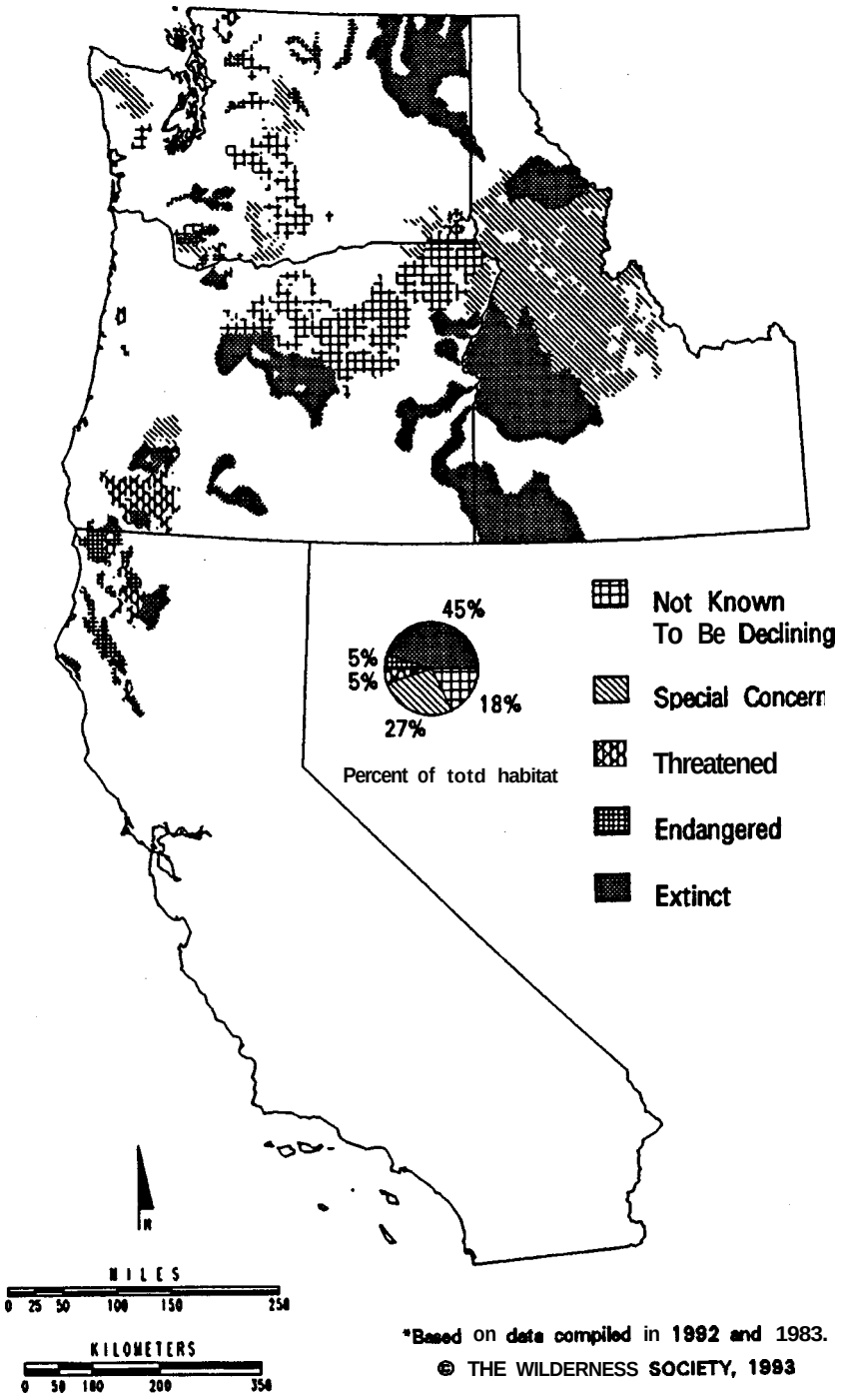


Figure 2-11. Status of summer steelhead in the Pacific Northwest and California. From TWS (1993). Reproduced with permission of the publisher.

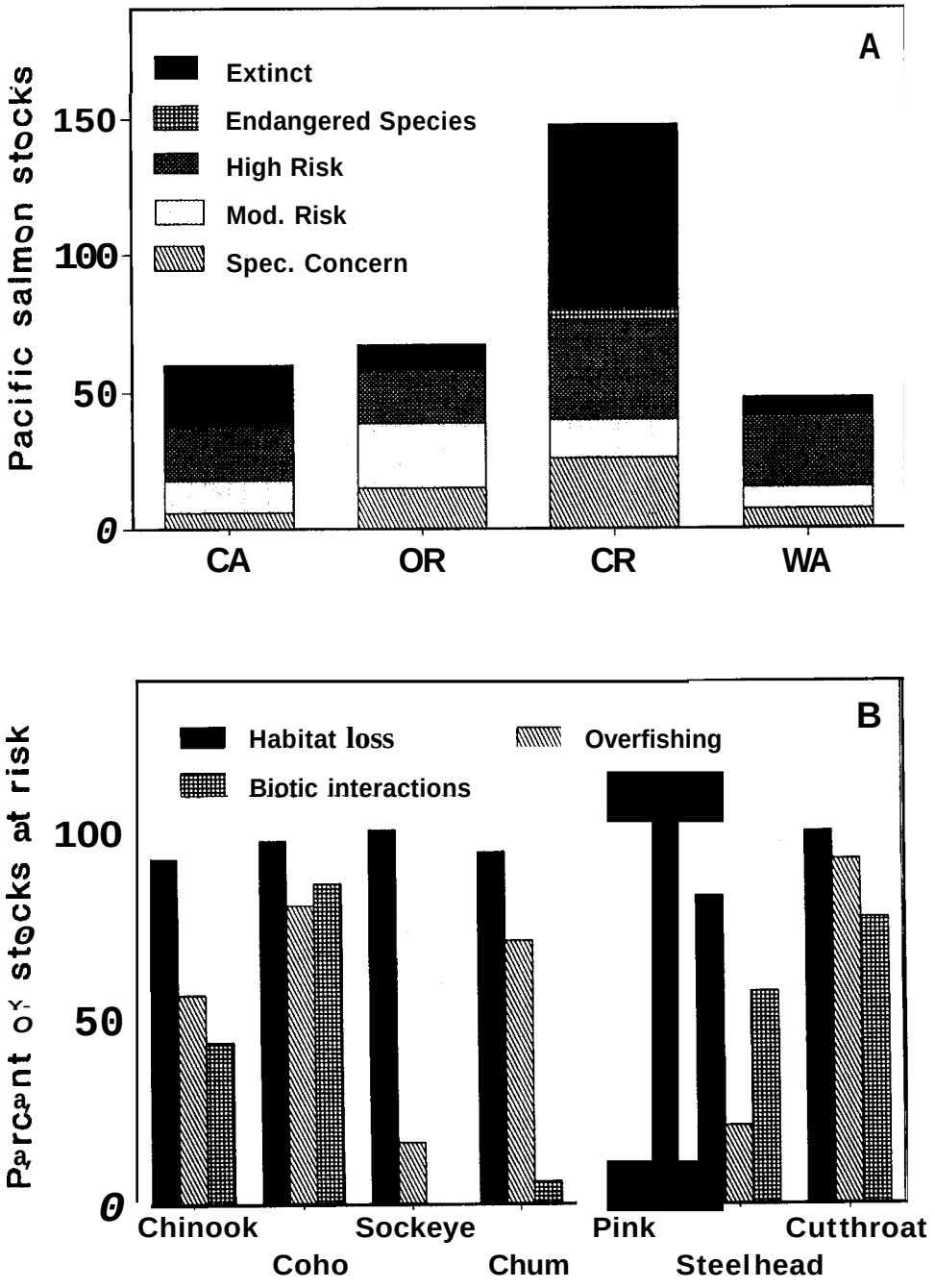


Figure 2-12. **(A)** Distribution of stocks of anadromous Pacific salmon *(Oncorhynchus)* in different extinction risk categories within various portions of the Pacific coast. (B) The percentage of stocks in which habitat damage, overfishing, and harmful biotic interactions have been implicated in declines **of** stock abundance. Figure from Bisson et al. (1992b) based on data from Nehlsen et al. (1991). Reproduced with permission of the publisher.

the Columbia Basin and along the coast (Figure 2-3). Today, coho stocks in the eastern half of their range are extinct, and stocks in the southern two-thirds of their coastal range are considered imperiled (Frissell 1993b). High numbers of threatened and endangered species in the Puget Sound and San Francisco Bay areas suggest that urbanization has contributed to the declines of native taxa.

In addition to the above reports, NMFS is now preparing status reviews of seven eastern Pacific anadromous salmonids over their ranges in the region. These reviews will incorporate information from the publications cited above as well as from state-wide status reviews prepared by the Oregon Department of Fish and Wildlife (in preparation) and the Washington Departments of Fisheries and Wildlife (WDF et al. 1993). These reviews will focus on delineating "evolutionarily significant units" (see Section 4.2.5) pursuant to potential listing as threatened or endangered under the Endangered Species Act, and they may differ somewhat from other reporting efforts.

Evidence of aquatic habitat degradation is not limited to salmonids. Counts of Pacific lamprey, *Lampetra tridentata,* at Winchester Dam on the Umpqua River in Oregon have declined from 37,000 in 1965 to 473 in 1993 (ODFW unpublished data, cited in Li et al. 1995), and lamprey returns to the Snake River numbered fewer than 20 (WDF unpublished data, cited in Li et al. 1995). Reductions in lamprey populations have likely resulted from a combination of habitat modification and the loss of salmonids as hosts; these losses demonstrate that declines are not restricted to fish species intensively harvested for consumption by humans. Amphibians, which use streams and wetlands as breeding and rearing habitats, are also highly sensitive to environmental degradation (Welsh 1990). Recent field studies in the Pacific Northwest indicate widespread declines of populations, reductions of ranges, and extinction of amphibians in forest and other ecosystems. Blaustein et al. (1994) identified habitat destruction as the major cause of amphibian losses but suggested that other factors may be important, including chemical pollution, acid precipitation, increased ultraviolet radiation, introduction of non-native species, pathogens, harvesting by humans, and natural population fluctuations.

2.3 Cumulative Effects

The widespread decline of salmonid **stocks** throughout much of the Pacific Northwest has resulted from the cumulative effects of water- and land-use practices, fish harvest, hatchery practices, and natural fluctuations in environmental conditions. The term "cumulative effects" has been used generally to describe the additive or synergistic effects of these practices on ecosystems. Another comprehensive definition of cumulative effects is provided by (Sidle **1989):** "changes to the environment caused by the interaction of natural ecosystem processes with the effects of land use, distributed through time and space, or both."

Because of the longitudinal nature of stream ecosystems, the accrual of effects is important along both spatial and temporal dimensions. Activities that take place in headwater streams influence the suitability of habitats in downstream reaches (e.g., temperature change, sediment input) and affect the response of ecosystem components to additional stresses. Similarly, activities that have occurred in the past may influence current habitat conditions through residual effects (e.g., alterations in channel morphology caused by splash dams, hydraulic mining, channelization, and revetments) and long-term, persistent effects (e.g., reduced woody debris recruitment; loss of nutrients from salmon carcasses). And finally, some activities have latent effects on aquatic systems—effects that are triggered by future environmental events (e.g., mass wasting of hill slopes, debris torrents, incision of stream channels).

In the context of conserving and restoring salmonids, the notion of cumulative effects has at least two important implications. First, individual actions that by themselves are relatively minor may be damaging when coupled with other actions that have occurred or may occur in a watershed. Historical and current patterns of land-use practices, as well as other factors, have a significant bearing on how salmonid populations will respond to further anthropogenic disturbances. Traditional management strategies that rely on site-specific analyses without regard for other activities that have occurred or are occurring within a watershed or region will generally fail to protect salmonid populations against cumulative effects. This premise underlies the development of watershed and ecosystem approaches to resource management. Second, regional declines in salmonid populations are the product of numerous incremental changes in the environment. It is thus reasonable to expect that recovery of salmonid populations will proceed in a similar fashion—through incremental improvements in habitat conditions. Few activities directed toward improving habitat are likely to have sudden and marked influences on salmonid populations, and in many cases we may be unable to detect any improvement at all amid the "noise" of natural variation in salmonid production, except over long time periods (Hall and Knight 1981). This suggests that we should temper our expectations of how rapidly ecosystem complexity and integrity can be restored (Bisson et al. 1992b). It also means that individual stakeholders

can and must play an active role in salmonid habitat restoration even if tangible benefits are slow to come.

Cumulative effects of human activities have resulted in a regional landscape—including the aquatic ecosystems contained therein—that is highly fragmented with few large expanses of land (i.e., whole watersheds or basins) that are relatively intact (Doppelt et al. 1993). Early settlement of the Pacific Northwest was concentrated along low-gradient streams and rivers on relatively gentle terrain that was suitable for farming and ranching. Larger waterways served as primary travel corridors for boats as well as logs that were felled in or transported to the riparian zone and floated to downriver ports, Snagging operations removed thousands of logs annually to facilitate this river traffic (Sedell and Luchessa 1982). Similarly, roads and railroads typically were laid out in valley bottoms adjacent to rivers because gentler grades made construction easier. Diking and removing brush were commonly employed to reduce flooding of lowland areas and to allow farming and construction of houses within the historical floodplain. Streams were channelized to facilitate rapid runoff of stormwaters from watersheds.

A *de facto* consequence of these (and other) activities and their cumulative effects on salmonid habitats is that many of the most pristine habitats that remain are in headwater streams, where human disturbance has been less severe (Doppelt et al. 1993; Frissell et al. 1993a; Henjum et al. 1994). This situation has led to a common misperception that headwater environments are the preferred habitats of salmonids. In reality, headwater streams generally do not contain the wide array of habitats that are necessary or desirable for all life-stages of salmonids or for different fish species that have varying habitat requirements (Sheldon 1988). It is generally believed that unconstrained, aggraded floodplain reaches were once highly productive habitats for some anadromous salmonids (Stanford and Ward 1992). For example, off-channel areas adjacent to larger rivers have been shown to be important rearing habitats for salmonids during high winter flood events (Tschaplinski and Hartman 1983).

Fragmentation of habitat and the resulting isolation of populations may affect the long-term viability of salmonid stocks (see Section 4.2.4). In addressing fragmentation and connectivity of habitats for the northern spotted owl, Thomas et al. (1990) outline several general principles that are equally applicable to salmonid conservation:

- Large blocks of habitat are preferable to small blocks.
- Patches of habitat that are close together are superior to those that are far apart.
- Contiguous blocks are preferable to fragmented habitats.
- Interconnected patches are better than isolated habitat patches, and corridors linking habitats function better when they resemble the preferred habitat of the target species.

Thus, essential goals of salmonid restoration should be to prevent further fragmentation of aquatic habitats, to improve connectivity between isolated habitat patches, and to protect and restore areas surrounding critical refugia from further degradation so as to allow for the expansion of existing populations.

2.4 Strategies for Salmonid Conservation

In the last twenty years, there has been a fundamental shift away from "single-species management" of salmonids toward more holistic watershed and ecosystem approaches that seek to conserve aquatic habitats by protecting processes operating throughout the watershed. The Federal agencies responsible for administering public lands have concluded that ecosystem management is essential for arresting further habitat degradation, maintaining habitats that are relatively intact, and aiding in the recovery of at-risk species of fish (FEMAT 1993; FS and BLM 1994b, 1994c). Several recent efforts that incorporate an ecosystem perspective include the Aquatic Conservation Strategy in FEMAT (1993), the Eastside Forests Scientific Society Panel Report (Henjum et al. 1994), and the PACFISH strategy (FS and BLM 1994b, 1994c). BLM's strategy for managing wetland and riparian areas recognizes that "entire watershed condition is an important component in assessing whether a riparian-wetland area is functioning properly" (Barrett et al. 1993). EPAs Environmental Monitoring and Assessment Program (EMAP) is also based on the concept that all ecosystems existing in the landscape are integrated components and that the condition of one component affects and is affected by the condition of the others (Messer et al. 1991; Paulsen and Linthurst 1994). NMFS's coast-wide status review of coho salmon (Weitkamp et al. 1995) and steelhead trout, as well as the imminent coast-wide reviews of sea-run cutthroat trout and chinook, pink, chum, and sockeye salmon (NMFS 1994), further reflect a more comprehensive approach to resource management.

The **FEMAT** and PACFISH approaches to aquatic resource conservation as well as other published conservation strategies (Moyle and Sato 1991; Doppelt et al. 1993; Frissell et al. 1993; Henjum et al. 1994; Bradbury et al. 1995) share two common elements. First, each of these strategies

recognizes the importance of identifying and protecting those habitats that retain the highest degree of integrity to serve as refugia and centers from which population expansions can occur. Second, they recognize that an effective conservation strategy must emphasize restoring ecological processes and function and must be organized at a watershed (or larger) scale. "Key Watersheds" identified by the FEMAT report, the PACFISH strategy, and the Eastside Forests Scientific Society Panel Report illustrate these concepts as applied to Federal lands.

The historical abundance of many salmonids in the Pacific Northwest was due in part to the diversity of life-history types that evolved to exploit a wide array of available habitats and that allowed temporal and spatial segregation of habitat use. In the diverse, geomorphically and tectonically unstable environments of the Pacific Northwest, well dispersed networks of locally adapted salmonids are believed to be necessary for species persistence (Frissell 1993a). This diversity enhances the ability of species to adapt to continually changing environmental conditions. Furthermore, the anadromous life-history pattern exposes these fish to a tremendous diversity of habitats, which may include small headwater streams for spawning; larger streams, lakes, or off-channel areas for rearing; still larger streams as migration corridors; and estuaries and oceans for primary growth phases. Resident trout, char, and whitefish may also spend portions of their life histories in streams and lakes of various sizes. The success of salmonid populations depends on the availability of high-quality habitats needed during each life stage.

Conservation of salmonids will require a comprehensive approach that addresses these spatial and temporal needs. Current strategies for managing Federal lands began this process, but because of the spatial distribution of Federal lands, protected watersheds presently tend to be concentrated in higher-elevation areas, forested watersheds, and headwater streams. The FEMAT report specifically cites the importance of nonfederal lands in an overall riparian conservation strategy, and Henjum et al. (1994) further stress the need to accommodate a wide variety of habitat types through the establishment of Aquatic Diversity Areas. A strategy for nonfederal lands should build upon existing conservation plans by re-establishing connectivity between habitats on Federal and nonfederal lands, and by working towards protection of habitats that are poorly represented in Federal ownership, particularly the lower-elevation streams and habitats for resident species, including nongame fishes. (Both the FEMAT and PACFISH approaches focus on anadromous salmonids.) A strategy for salmonid conservation should also provide guidance for managers so that actions at a local scale can be integrated into watershed and regional recovery plans.

Local habitat rehabilitation is essential within this broader context of conserving habitats and biodiversity across broad landscapes. Improved land-use practices and rehabilitation of riparian zones can provide many benefits, including decreased sediment transport to the stream, decreased stream temperatures, increased allochthonous nutrient inputs, increased flood-plain interaction, stabilized ground water discharge, and increased inputs of large woody debris (Naiman 1992). As natural processes and conditions are restored, downstream reaches will be improved and connections between habitats re-established (Salo and Cundy 1987), allowing greater expression of life-history diversity. Thus, private landowners can play a vital role in both improving local conditions and advancing the recovery of salmonids region wide. Furthermore, local actions can enhance other values, including water quality and quantity.

2.5 What is Ecosystem Management?

The preceding section identifies several Federal and nonfederal programs or strategies intended to foster ecosystem management as it relates to aquatic systems. A recent study by the Congressional Research Service (1994) identified no fewer than eighteen Federal agencies that have committed to principles of ecosystem management, and various state and local government and nongovernment entities have made similar commitments (Christensen et al. 1996). Yet despite the apparent widespread acceptance of ecosystem management as a paradigm, the term "ecosystem management" can be taken to mean different things by different people (GAO 1994), and some people consider the term vague or imprecise. Many definitions found in the literature have common elements, such as "sustainability" or emphasis on protection of "ecological processes or functions," but without rigorous definition, these phrases too can be considered nebulous, opening the door for misuse or misinterpretation.

The Ecological Society of America Committee on the Scientific Basis for Ecosystem Management recently reviewed and synthesized much of the existing literature related to ecosystem management (Christensen et al. 1996). They identified eight essential components of ecosystem management including 1) sustainable management of resources, 2) clearly defined and operational management goals, 3) management based on the best available science and models, 4) recognition of the complexity and interconnectedness of ecological systems, 5) recognition that ecosystems are constantly changing, 6) acknowledgement that ecosystem processes operate

at multiple temporal and spatial scales, 7) the need to consider humans as integral parts of ecosystems, and 8) the importance of adaptability and accountability in management (Table 2-2). A key aspect of their definition is that "sustainability" is applied not to specific goods or services that ecosystems provide but rather to the ecological processes and structures that give rise to these goods or services.

We concur with Christensen et al. (1996) that these components form a sound basis for ecosystem management, and readers of *An Ecosystem Approach to Salmonid conservation* will find discussion of each of these elements throughout the document. Those elements pertaining primarily to ecological processes (e.g., sustainability of resources and ecosystem processes, complexity and interconnectedness of ecosystems, temporal and spatial aspects of ecological processes) are addressed mainly in Part I; aspects related to implementation of ecosystem management (e.g., management goals, social dimensions, adaptive management) are discussed in Part II. Our purpose in highlighting these elements of ecosystem management in this section is to provide a frame of reference from which to organize material presented in the remaining chapters.

Table 2-2. Essential components of ecosystem management. Based on recommendations of Ecological Society of America (1995).

Attribute	Description
Sustainability	Ecosystem management entails managing in such a way as to ensure that opportunities and resources for future generations are not diminished. Sustainability should not be evaluated based on the delivery of specific goods and services, but rather on the maintenance of the ecosystem structures and processes necessary to provide those goods and services.
Goals	Ecosystem management requires clearly defined goals. These goals should not focus exclusively on individual commodities (e.g. board feet of timber, catch of fish, visitor days). They should be explicit in terms of desired future trajectories or behaviors for components and processes necessary for sustainability.
Sound ecological models and understanding	Ecosystem management is founded on sound ecological principles, emphasizing the role of ecosystem structures and processes. It must be based on the best science and models currently available.
Complexity and connectedness	Ecosystem management recognizes that ecological processes are complex and interwoven and that this complexity and connectedness may confer particular properties (e.g., stability, resistance, resilience) to ecosystems.
Recognition of dynamic nature of ecosystems	Ecosystem management recognizes that environmental change and biological evolution are inherent properties of ecosystems and that attempts to maintain particular ecosystem "states," rather than ecological capacities, are futile over the long term in a changing environment.
Context and scale	Ecosystem management acknowledges that ecosystem processes operate over a wide range of spatial and temporal scales and that their behavior (including their response to human perturbations) at a given location is strongly influenced by the surrounding landscape or system and by the legacy of past events.
Humans as ecosystem components	Ecosystem management acknowledges that humans are components of ecosystems, as well as the source of most significant challenges to sustainability. Humans who are a part ecosystems will, of necessity, define the future of those ecosystems. Thus, ecosystem management applied alone, without consideration of social and economic systems (and their sustainability), is insufficient to ensure resource sustainability.
Adaptability and accountability	Ecosystem management recognizes that current models and paradigms of ecosystem structure and function are provisional and subject to change. Acknowledging limits to scientific understanding and adapting to new information as it becomes available are central to successful ecosystem management.

3 Physical and Chemical Processes

The freshwater and estuarine habitats of salmonids are the product of interactions among numerous physical, chemical, and biological processes (Marcus et al. 1990; Swanston 1991) operating over long- and short-term temporal scales as well as large and small spatial scales. Over millions of years, tectonic and volcanic activity in the Pacific Northwest has created a region of extreme topographic complexity, characterized by a series of mountain ranges that are oriented along a north-to-south axis and separated from one another by lowlands, plateaus, or smaller mountain ranges. Significant portions of the Pacific Northwest landscape have been reshaped by glacial advance and recession. These large-scale, long-term, geomorphic and climatic processes have created the physical template upon which rivers and estuarine systems of the Pacific Northwest have formed.

Within a watershed, topographic, geologic, and climatic characteristics control soil development and vegetation cover as well as influence the transport of water, sediments, wood, and dissolved materials from upland areas to the stream channel. These transport processes occur continuously but may be greatly accelerated during natural disturbances such as floods, debris torrents, landslides, and wildfires. The riparian zone acts as a filter that moderates the exchange of materials from terrestrial to aquatic ecosystems. In addition, riparian vegetation directly controls stream environments by providing shade and stabilizing streambanks and through the input of organic litter and large woody debris.

A useful way to conceptualize how these processes ultimately affect salmonid habitats is in terms of a hierarchy of factors (Frissell et al. 1986; Naiman et al. 1992), where each component exerts influence on other components—usually at the same or lower levels—and all components ultimately influence the character of the stream, lake, or estuary (Figure 3-1). Elements at the top of the hierarchy (e.g., climate, geology, topography, soils, and vegetation) have pervasive effects on other processes occurring in a basin or watershed (e.g., sediment delivery, hydrology, nutrient cycling,

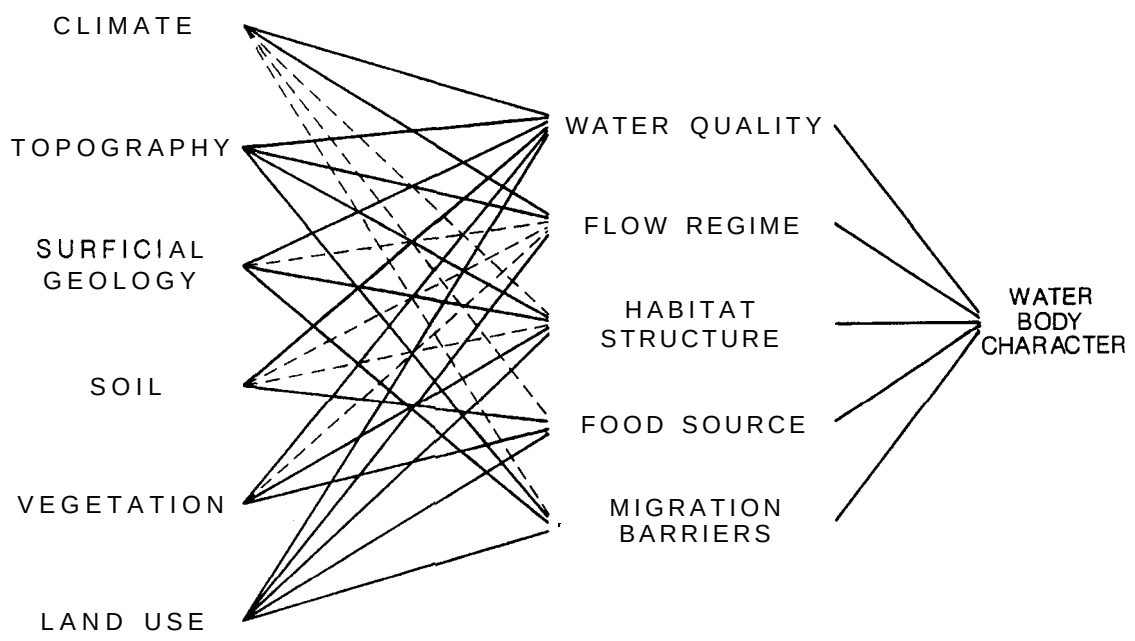


Figure 3-1. The influence of watershed characteristics on the character of aquatic ecosystems. Solid and dotted lines represent greater and lesser influences, respectively. Modified from Hughes et al. (1986). Reproduced with permission from the publisher.

riparian features) that give rise to the water body characteristics (e.g., water quality, flow regime, habitat structure, aquatic biota). Lower-tier processes generally interact with other components at the same and lower levels but may also influence components at higher levels through feedback loops. Specific characteristics of salmonid habitats are thus the manifestation of highly complex interactions among processes operating over many spatial and temporal scales.

In this chapter, we provide a broad overview of the dominant physical and chemical processes affecting the landscape and, ultimately, the aquatic ecosystems on which salmonids depend. The relative influence exerted by each specific process varies across the landscape with differences in geomorphology, geology, climate, hydrology, soil, vegetation, and other controlling factors. Consequently, the potential effects of human disturbances on aquatic systems are similarly variable in space. Our objective is to provide sufficient detail of physical and chemical processes so that regional differences in the response of ecosystems to human-caused perturbations can be understood. We begin with a review of processes that operate over large temporal and spatial scales and over which humans have minimal influence. Next we review processes that operate at smaller spatial and temporal scales and that may be substantially altered by land-use activities. Included in this discussion is a review of functional roles of riparian vegetation with respect to salmonid habitats. A summary of the effects of physical and chemical processes on salmonids and their habitats is presented in Section 3.10.

3.1 Tectonism and Volcanism

Tectonic activity operating over millions of years created the rugged montane physiography, high local relief, and steep slopes of the Pacific Northwest—structural features that control the geographic patterns of drainage systems in the region. These processes set the stage for other geomorphic processes that shape stream channels.

Direct effects of tectonics on active geomorphic processes generally are limited in spatial extent and relatively infrequent, compared to other processes discussed in this document. The Pacific Northwest is subject to large subduction zone earthquakes at intervals of severai hundred years. These large-magnitude earthquakes may cause subsidence in soft alluvial and coastal fills, creating zones of deposition (Atwater 1987; Darienzo and Peterson 1990), and they may also trigger mass movements of soil.

Volcanic activity has been less significant regionally than tectonics and glacial processes, but at local sites it has resulted in catastrophic readjustments of the landscape. Geomorphic impacts depend on the geochemical type of volcanism. Explosive eruptions of silicic volcanoes directly reshape the landscape, blocking and diverting drainage systems by ash flows, filling valleys or channels with mudflows, and causing major inputs of sand and silt-sized sediments from tephra (airborne ash). Basaltic volcanic centers may block and divert drainage systems through lava flows and cinder eruptions and also release limited amounts of tephra. Recently active silicic volcanic centers are limited to the Cascade Range (Sarna-Wojcicki et al. 1983). Basaltic volcanic centers active in the Holocene are found in the Cascades from southern Washington to California, a few areas of eastern Oregon, and in the eastern Snake River Plain of Idaho.

Volcanic mud- and ashflows commonly occur from volcanic eruptions in Cascade Range volcanoes. Mudflows have produced the most widespread geomorphic effects of past eruptions and can inundate valley floors with deposits less than one meter to tens of meters thick. Mudflows caused by the 1980 eruption of Mount St. Helens inundated valleys and completely buried pre-existing river channels. Channels subsequently re-established on the mudflow deposits through alternating episodes of incision, channel widening, and aggradation over a period of at least several years (Meyer and Martinson 1989). During the adjustment period, sediment yields were much higher than before the eruption. Geomorphologic adjustments have been prolonged by landslides on slopes that were destabilized by the eruption. Ash flows also move down valleys and bury valley floors (Crandell 1976), while tephra may be deposited many kilometers from the source.

3.2 Glaciation

The landscape of the Pacific Northwest has developed under alternating glacial and interglacial periods over the last one million years or longer. Glaciation has affected the region's landscapes through 1) direct modification of mountain areas and limited lowland areas by glaciers; ***2)*** eustatic sea-level lowering, which has had major effects on coastal rivers and estuaries; **3)** glacial-interglacial climatic changes that have influenced the hydrologic regime; and **4)** climate-driven changes in vegetation cover that have affected hillslope and stream processes (Table **3-1).** In general terms, glacial periods are times of rapid sediment transfer from uplands to lowlands and to the ocean by glacial advance and meltwater transport in glaciated areas and by increased streamflow in unglaciated areas. Interglacial periods tend to be periods of sediment accumulation in upland valleys with limited fluvial transfer out of the uplands (Thorson 1987).

Table 3-1. Past controls and effects on landscape development in the Pacific Northwest.

Period	Controls	Probable geomorphic and ecological effects
Glacial	In glaciated and periglacial areas:	
	Advance of Cordilleran ice sheet; development of mountain ice sheets and alpine glaciers; very cold climate with reduced precipitation.	Glacial erosion and deposition and formation of outwash trains in valleys; periglacial churning and mass movement, intensified mechanical weathering; glacial meltwater discharge; displacement of interglacial ecological communities; vegetation cover absent or greatly reduced.
	In unglaciated areas:	
	Lowered sea level; cold climate with reduced precipitation.	Displacement and shrinkage of estuary areas; reduced vegetation cover; mechanical weathering, mass movement and slope erosion rates greater than modern; increased streamflow and fluvial sediment transport; accumulation of coarse valley fills; reduced organic inputs to streams.
Late Glacial and early Holocene	Retreat and downwasting of glaciers; rapid sea-level rise; warming; effective moisture greater than modern ca. 14,000 to 11,000 years ago, then less than modern until ca. 7,000 years ago.	Glacial deposition and exposure of glaciated land surfaces; landward displacement of estuaries, increase in estuary depth and area; mass movement and slope erosion rates decreasing but still greater than modern; streamflows probably greater than modern; stabilization and then incision of valley fills; increasing vegetation cover and changes in community composition; increased organic inputs to streams, but still less than modern; minor fluctuations in alpine glaciers.
Middle Holocene to modern	Sea-level stable; climate approaching modern conditions, with short-term fluctuations.	Estuaries filling and shallows developing; slope stabilization and decrease in mass movement rates; decreased mechanical and increased chemical weathering; streamflows near modern, with short-term fluctuations; continued but slowed incision of valley fills; development of modern ecological communities; high rates of organic inputs to streams; minor fluctuations in alpine glaciers.

Where residence times of sediment accumulations or recurrence intervals of events are thousands of years (Dietrich et al. 1982; Kelsey 1982), glacial-interglacial transitions may be the most important periods in landscape formation.

During the last glacial period, about 22,000 to 15,000 years ago, ice sheets and mountain glaciers were developed in many areas of the Pacific Northwest; sea level was about 100 m below present, exposing large areas of the continental shelf. The Cordilleran ice sheet extended south from British Columbia, covering the Puget Lowland, northern Cascades, Okanogan Valley, and upper Columbia Valley in Washington. South of the ice sheet, mountain ice sheets and glaciers were widely distributed in the mountainous regions of Washington, Oregon, Idaho, and the Siskiyous of Northern California (Crandell 1965; Porter et al. 1982). Climate of the glacial period was much colder than today. Although effective moisture in the Pacific Northwest was less (Thompson et al. 1993), runoff likely was as high or higher than today, because of changed land-surface conditions. Down the valley from glaciers and in unglaciated watersheds, frost weathering and mass wasting were probably more intense than at present. River systems probably had greater streamflow and transported greater sediment loads. In addition, enormous ice jams periodically

developed and broke, resulting in catastrophic floods that formed the coulees of eastern Washington and deposited the deep soils of the Willamette Valley.

In addition to these physical changes, ecological changes resulting from glacial climates may have also influenced geomorphic processes. Preglacial ecosystems of these areas were significantly displaced by glaciation; species and stocks present today in these ecosystems likely existed in refugia south of or at lower elevations than the glaciers. In unglaciated parts of western Oregon and Washington, the vegetation consisted of tundra close to glaciers and subalpine parkland elsewhere, including the Oregon Coast Range (Worona and Whitlock 1995). In eastern Washington, a sparse periglacial steppe was present (Barnosky et al. 1987; Whitlock 1992; Thompson et al. 1993). In the more sparsely vegetated landscape of the last glacial period, less large organic debris was available to influence streams and valley floors. Present environmental conditions have prevailed in this region for the last 6,000 to 8,000 years. Current conifer forest communities did not become established until 5,000 to 2,000 years ago (Whitlock 1992; Worona and Whitlock 1995), and in glaciated watersheds of western Washington, stream channels reached conditions similar to those of the present by about 6,000 to 8,000 years ago (Benda et al. 1994). As density and height of forest stands increased with climatic amelioration, woody debris exerted a stronger influence on stream and valley morphology. Some channel incision and narrowing of meander belts probably continued into the late Holocene.

A general model of river channels based on empirical evidence from several parts of the world suggests that channel changes from glacial to interglacial periods follow a specific sequence. Braided channels dominate during glacial periods. During interglacial periods these change to transitional, braided, meandering channels with mid-channel bars but well-defined thalwegs, and then to large meandering channels adjusted to higher-than-present discharge. Finally, smaller meandering channels develop during stable conditions typical of post-glacial periods (e.g., late Holocene; Schumm and Brakenridge 1987). In the Pacific Northwest, the late-glacial to early Holocene period was likely characterized by channel incision into thick glacial-period valley fills, formation of terraces, sediment yields higher than present as rivers downcut, and significant changes in channel morphology because of changed hydrologic and sediment regimes (Benda et al. 1994).

In addition to the changes in inland watersheds described above, coastal rivers were directly affected by lowered sea level during glacial periods (McDowell 1987). At the last glacial maximum, global sea level was 100 m or more below the present sea level, and the shore was 10 km or more west of its present location. Coastal streams flowed across the exposed continental shelf, perhaps in incised valleys. Estuaries were very limited in extent. As global deglaciation began, sea level initially rose very rapidly creating deep coastal estuaries. Beginning 10,000 years ago, the rising sea level continued at a decreasing rate, and it has fluctuated close to the present level since 4,000 years ago. Shallow-water conditions in estuaries, including mud and sand flats, have become established only recently (McDowell 1986, 1987).

3.3 Wildfires

The historical frequency of fires varies over the landscape as a function of climate and vegetation type. Fires in high-elevation communities of subalpine fir, western hemlock/red cedar, lodgepole pine, and grand fir tend to recur at an interval of decades to centuries; low- to mid-elevation juniper, ponderosa pine, Douglas-fir, and white fir forests typically experience fires at intervals of several years to a few decades. Little information is available about the historical frequency of fire in grassland, shrubland, and woodland communities east of the Cascade Crest (Agee 1994). Plant assemblages of stiff sagebrush and Sandberg's bluegrass may have biomass sufficiently low to prevent large-scale fires, while other communities, including various fescue and bluebunch wheatgrass assemblages, may have sufficient biomass to carry fire but lack sources of ignition (i.e., lightning) during the periods when they are most combustible (Agee 1994). Fire frequency in other sagebrush and woodland communities is poorly documented. In the Cascade region, wildfire regimes are highly variable. Morrison and Swanson (1990) reconstructed fire histories at two locations in the central and western Cascades and estimated recurrence intervals of approximately 95 and 149 years, respectively (range 20–400 years); most fire-created patches were less than 10 hectares in size. In the Coast Range, higher humidity, more lush vegetation, and less frequent lightning storms combine to reduce the frequency of wildfire; however, under dry summertime conditions, the effects of wildfire in dense timber stands can be substantial. During the period 1933–1951, four fires in the Coast Range of Oregon, collectively known as the Tillamook fires, burned more than 260,324 hectares (643,000 acres) and had significant and long-lasting effects on forest and riparian communities. Although these fires were human-caused, they demonstrate the potential for forests in the Coast Range to burn under certain circumstances.

Riparian areas generally are characterized by a higher percentage of deciduous plants than is found in surrounding uplands. In addition, local

microclimates tend to be cooler, resulting in moist soils and high fuel moisture, especially in floodplain woodlands. Because of these attributes, riparian areas do not burn, or they burn at lower intensity than forests in upland areas. As such, they may buffer aquatic communities from some of the effects of wildfire. However, in headwater reaches and at higher elevations, stronger winds and greater biomass may facilitate fires of relatively high intensity. Consequently, it is difficult to generalize about the effects of fires on the riparian zone (Agee 1994).

Fires in upland areas and riparian zones can affect aquatic ecosystems by altering vegetation cover, which in turn influences erosion and sediment transport, water infiltration and routing, the quantity of nutrients reaching streams, the amount of shading, and the input of large woody debris into the system (Wissmar et al. 1994). The extent of impacts is generally related to the intensity of the burn. In high intensity fires, soil organic matter that helps hold soils together is consumed, increasing the susceptibility of soils to erosive forces. In addition, volatilization of certain compounds can cause the surface soil layer to become hydrophobic, thereby reducing infiltration of water and increasing surface runoff (Marcus et al. 1990). The combined effects of vegetation loss and hydrologic changes can alter the frequency of severe debris torrents (Wissmar et al. 1994). Nutrients such as phosphorous, nitrogen, and sulfur may be volatilized into the atmosphere (Everest and Harr 1982) or lost through leaching and soil erosion. The loss of riparian vegetation can increase exposure to solar radiation, causing streams to warm. Inputs of large woody debris may also change following fire in the riparian zone. In speculating about the effects of the Yellowstone fire of 1988, Minshall et al. (1989) hypothesize that large woody debris in streams would likely increase immediately following the fire—from augmentation of existing woody debris with falling branches—then decrease through time because new growth contributes little to instream woody debris.

Humans have significantly altered natural fire regimes through land-use practices and an extensive and long-term focus on fire suppression. As a result, significant changes in forest vegetation have resulted. East of the Cascades, fire suppression has led to shifts in vegetation from historically open stands of ponderosa pines and western larch to stands with dense understories of Douglas-fir and grand fir (Mutch et al. 1993). Ponderosa pines are well adapted to frequent, low-intensity burns that were characteristic of eastside forests. These fires tended to prevent fire intolerant species from invading. Drought and subsequent insect infestations have killed many understory trees, allowing fuels to accumulate and increasing the probability of high intensity fires (Wissmar et al. 1994). Consequently, ecosystems that once experienced frequent but small wildfire disturbances are now prone to infrequent but much more catastrophic events.

3.4 Sediment Transport

Sediment transported from upland areas into stream channels determines the nature and quality of salmonid habitat in streams, rivers, and estuaries. The development and persistence of morphological structures used for spawning, incubation, and rearing depend on the rate at which sediment is delivered and the composition of deposited materials. Sediment delivery rates and composition, in turn, are controlled by climate, topography, geology, vegetation, and hydrology. Local variation in these watershed characteristics ultimately determine the type and quality of habitat found in a given system.

Land-use practices, through alteration of soil structure, vegetation, and hydrology, can significantly alter the delivery of fine and coarse sediments to streams, thereby affecting salmonid habitats. In this section, adapted primarily from Swanston (1991), we discuss surface erosion and mass wasting, the dominant forms of sediment transport, as well as environmental factors that influence these processes. The routing of sediments within the stream channel and the role of large woody debris in controlling sediment movement are discussed in Sections 3.5 and 3.9.5 of this document.

3.4.1 Surface Erosion

Surface erosion results from rain and overland runoff. Particulate and aggregate materials are relocated via a two-step process: detachment then downslope transport of detached materials. Detachment is influenced by the size and compaction of particles and by the protective cover of organic litter and plants. Slope gradient and length, rainfall intensity, and soil infiltration rate determine transport rate (Swanston 1991). Initiation of surface erosion may be caused by landslides, fire, logging, rain, drop splash from forest overstory, animal activity, freeze-thaw phenomena, or any other surface disturbance of soil. Surface erosion rarely occurs on undisturbed forest lands west of the Cascade crest because of high infiltration rates, though it may occur in areas with steep ($> 27°$) slope gradients (Swanson et al. 1987). In sparsely vegetated lands east of the Cascades, the potential for surface erosion is greater because of the lack of groundcover.

Most surface sediments that reach stream channels result from channelized erosion (rilling and gullying) and sheet erosion (Brown 1980; Swanston 1991). Channelized erosion occurs when flows are concentrated and restricted by landforms, usually following heavy storms or snowmelt (Beschta et al.

1995). It is considered the most significant form of surface erosion on forest lands (Brown 1980). Although uncommon in undisturbed forested situations, rills may occur when infiltration capacity is reduced (Megahan 1991). In contrast, nonchannelized erosion develops from detachment begun by raindrop-splash and overland flow (sheet erosion) or by gravitational and wind movement of dry particles (dry ravel). These processes generally occur on exposed soils and tend to remove soil uniformly over an exposed area. Sheet erosion tends to be of greater significance on low-gradient agricultural lands than on forested lands, whereas dry ravel occurs on steep slopes in soils lacking cohesion (Swanson et al. 1987; MacDonald and Ritland 1989).

3.4.2 Mass Wasting

Mass wasting—slumps, earthflows, landslides (or debris avalanches), and soil creep—is often a major component of sediment delivery to streams, particularly in mountainous regions where surface erosion is minor (Swanson and Dyrness 1975). Generally episodic in nature, mass wasting can provide large quantities of sediment and organic material to streams. Reeves et al. (1995) suggest that historically in the Coast Range, these periodic natural disturbances (sometimes associated with wildfire) served to replenish large woody and coarse sediment in streams at intervals ranging from decades to several centuries or more. Following these disturbances, natural erosion and aggradation processes gradually modified these disturbed reaches, causing a succession of different habitat conditions for salmonids. This variation in space and time created areas of naturally excellent and poor salmonid habitat. By increasing the frequency (both spatial and temporal) and altering the nature of these disturbances (e.g., reducing the quantity of large woody debris associated with mass failures), humans have degraded and simplified stream habitats.

Slumps and earthflows generally develop in deeply weathered soils. These often occur in sedimentary geology (siltstones, sandstones, mudstones) and volcaniclastic rocks, In soils with primarily clay-sized particles, low soil permeability restricts groundwater movement and causes puddling and fluid soils (Swanston 1991). These unstable soils produce slumps and earthflows. Slumps are the sliding of soil blocks along a concave surface, and earthflows often begin as slumps or a series of slumps. Once initiated, rheological flow of the clay fraction keeps the individual soil **blocks** moving downslope like a viscous fluid in earthflows. Earthflows tend to be seasonal with most movement occurring after heavy rains have saturated soils. These flows are slow moving, ranging from 2.5–2,720 $cm \cdot yr^{-1}$ (Swanston 1991) and may eventually protrude into the stream channel, where they are gradually eroded away. As they erode, residual lag deposits may form, which can increase channel gradient downstream through the accumulation zone. These areas, if in otherwise "sediment poor" reaches, and if they contain coarse sediments, may increase the habitat diversity in a morphologically uniform channel and have a long-term beneficial effect on fish habitat.

Soil creep is soil movement that is imperceptible except by measurements taken over long periods of time. Carson and Kirkby (1972) identify causes including reworking of the surface soil layers because of frost heaving, steady application of downward sheer stress, and random movements from organisms or microseisms. Continuous creep tends to occur in clay soils and is absent in coarse-grained soils.

Landslides typically occur in shallow noncohesive soils on steep slopes overlying less permeable bedrock (Beschta et al. 1995). Conditions causing landslides include 1) zones of weakness in soil or bedrock, 2) wind stress transferred to the soil by trees, 3) deformation caused by soil creep, 4) drag caused by seepage pressure, and 5) removal of slope support by undercutting. Landslides—relatively dry soil masses—are distinguished from debris flows, which are typically saturated. When landslides enter stream channels during floods, they become debris flows—large volumes of water containing soil, rock, and, frequently, large organic debris. These flows scour the channel and severely modify fish habitat as they move rapidly downstream. As debris flows move downstream into higher order channels, their effects become less pronounced because of increasing streamflow.

3.4.3 Factors Affecting Erosion and Sedimentation Rates

The magnitude, locations, and frequency of sediment delivery to active channels is highly dependent upon climate, local topography, soil type, soil saturation, vegetative cover, organic matter, depth and degree of weathering, and degree of upslope disturbance (Swanston 1991; Beschta et al. 1995; OWRRI 1995). Rain-dominated watersheds tend to yield more sediment than snow-dominated systems, although interbasin variability is quite high. Larson and Sidle (1981) examined data from 13 relatively undisturbed watersheds and reported sediment yields of 2.0 to 40.7 $tonnes \cdot km^{-2} \cdot yr^{-1}$ for rain-dominated systems. For snow-dominated systems, sediment yield typically ranged from **1.6** to 6.1 $tonnes \cdot km^{-2} \cdot yr^{-1}$; however, two watersheds had substantially higher yields of 39.9 and 117.1 tonnes. $km^{-2} \cdot yr^{-1}$ (see Swanston 1991). Within-year variation in sediment production can also be high. Larson and Sidle (1981) reported differences in sediment yield

among years of an order of magnitude or more for both rain-dominated and snow-dominated systems.

The timing, frequency, and type of precipitation influences the rate and yield of sediment delivered to stream channels. In rain-dominated regions, sedimentation and allochthonous inputs are minimized during summer low-flow periods. Sedimentation increases during the wet months of September to February when soils are saturated and landslide hazards are highest. In snow-dominated regions, sedimentation is greatest during periods of rapid snowmelt or during high-intensity rain storms, when high streamflows occur and entire hillslope and channel systems erode (Swanston 1991).

Topography influences slope steepness, length, elevation, and aspect. Runoff energy is highest on steeper slopes with greater slope length, which increase the volume and velocity of water moving downslope. Failures that occur on lower areas of the hillside nearer streams have a greater potential of reaching the stream.

Parent material and soil types also determine soil texture and erodibility. Erodible soils include those derived from granite, quartz diorite, granodiorite, Cenozoic nonmarine sediments, and schist (Beschta et al. 1995). Diorite and various metamorphic, rock-derived soils have intermediate erodibility, and nonerodible materials include andesite, basalt, peridotite, serpentinite, and pre-Cenozoic and Cenozoic marine sediments. Important soil properties affecting mass wasting include cohesion, structure, porosity, moisture capacity, drainage, chemical properties, and soil depth, all of which are affected by the relative proportion of clay, silt, and sand in the soil (Swanston et al. 1980). Typically, soils with little cohesion, structure, or porosity, low moisture capacity, and poor drainage are more likely to erode.

Vegetative cover tends to reduce sediment transport by reducing detachment rate and through the binding capacity of root masses (Larson and Sidle 1981; Harvey et al. 1994). Organic matter, utilizing water as the cementing agent, helps to form aggregates that tend to be more resistant to detachment and transport (Dyrness 1967).

3.4.4 Regional Differences

East of the Cascades, soils are most susceptible to surface erosion, but mass wasting events can be locally important. Slumps originate in fine textured soils, while debris-torrent failures occur in weakly cohesive ash Harvey et al. (1994) suggest that the high infiltration rates in most soils of the inland Pacific Northwest region make them less susceptible to surface erosion unless slopes are greater than 30% and not vegetated. Compacted ash and pumice soils on shallow slopes are susceptible to gullying because of their low density and cohesion. In areas characterized by coarse, cohesionless soils and periods of drought, dry creep and sliding of materials from denuded slopes may be an important source of local surface erosion (Swanston 1991). Dry ravel is significant on slope gradients greater than 22" in pumiceous, cindery, and ashy soils—conditions found in specific areas in the central Oregon plateau and eastside portions of the Cascade Range in Washington. Dry ravel is also common in dryer parts of Idaho, southwestern Oregon, and the Cascades in Oregon (Swanson et al. 1987).

Mass wasting occurs with high frequency in the western Cascade Mountains and Coast Range (MacDonald and Ritland 1989; Beschta et al. 1995). Wet climatic conditions in the Coast Range and valleys tend to promote deep soil formation and clays, which are prone to slow continuous failures, including slumps, soil creep and earthflows. At higher elevations near the Cascade crest, shallow, cohesionless soils overlying slightly weathered bedrock are susceptible to landslides. Sediment budgets from three sites illustrate these regional differences. The wet, snowmelt-dominated, glaciated, and tectonically active Queen Charlotte Islands of British Columbia have sediment yields an order of magnitude greater than drier, snowmelt- dominated, granitic lands of central Idaho, while in the rain-dominated regions of western Oregon and Washington, yields are intermediate to the other two regions (MacDonald and Ritland 1989).

3.5 Channel Morphological Features and Their Formation

Stream conditions important for aquatic habitat can be observed over a range of scales from an entire drainage network to a reach to a channel unit (Gregory et al. 1991). Average values of many stream characteristics, such as width, depth, velocity, and bed material size, vary systematically in a downstream direction, There are, however, important patterns of variation at local scales, such as the reach and the channel unit scales. Reaches are stream and valley segments, typically 1–10 km long, within which gradient, valley width, and channel morphology are relatively homogeneous and distinct from adjacent segments. Reach-scale variation is controlled by geologic factors such as rock type, geologic structure, and location of geomorphic features such as terraces, alluvial fans, and landslides (Table ***3-2)***. In many streams, high-gradient reaches with narrow, constrained valley floors are interspersed with lower-gradient, alluvial reaches with wide valley floors (Grant et al. 1994).

Table 3-2. Reach classes in small Oregon streams. From Frissell et al. 1986. Reprinted with permission from the publisher.

Gross typology	Morpho-genetic class*	Morpho-genetic process	Relative length	Mean slope†	Dominant substrates	Develop-mental trend	Potential persistence‡
EROSIONAL	Bedrock outcrop	Irregular bedrock; resistance to weathering	Moderate to short	Variable: moderate to steep	Bedrock	Stable; all sediments transported	Long term
(Zones of exposure of bedrock floor or trend toward degradation of bed)	Colluvium (nickpoint)	Downcutting through landslide or torrent debris	Moderate to short	Steep, later becoming moderate	Boulders, cobbles, clay soil	Active degradation (unless reloaded	Generally moderate; depends on deposit size
	Torrent scour	Channel scour by debris torrent or flood	Moderate to long	Moderate to steep	Bedrock, some boulders	Transport of most sediments; local aggradation	Moderate (due likely to recruitment of constructional features
Channel pattern: straight	Alluvium	Downcutting through alluvium of old constructional reach	Moderate	Moderate	Cobbles, gravels	Slow degradation	Moderate to short term
	Root blockage	Channel shift after colluvium or debris jam blockage; tree roots delay downcutting	Short to moderate	Moderate to low	Tree roots, gravels, cobbles, clay soil	Stable period followed by degradation	Short term; very short if small roots
CONSTRUC-TIONAL	Bedrock outcrop	Sediment storage behind resistant bedrock features	Variable	Low	Gravels, fines, bedrock	Stable; inputs balance outputs	Long term
(Zones of aggradation and alluvium)	Colluvium	Sediment storage behind landslide or debris torrent deposits	Variable	Low	Gravels, cobbles, fines	Degradation, shortening (unless reloaded)	Long term to moderate (depends on deposit size)
Channel pattern: straight often verging on braided	Large woody debris	Sediment storage behind large logs or debris jams	Moderate	Low	Gravels, fines, wood	Net aggradation until decay or washout	Moderate, sometimes long term
	Small woody debris	Sediment storage behind jam of small debris	Short	Low to moderate	Gravels, cobbles, fines, wood	Aggradation, then quick washout	Short term

* Morphogenetic classes are further subdivided by segment class, whether banks are clayey colluvium or gravelly alluvium, whether sideslopes allow lateral migration, and by riparian vegetation state.

† Slope scale: moderate = same as segment slope, low = less than segment slope, and steep = greater than segment slope.

‡ Persistance scale: long term = > 100 years, moderate = 20–100 years, and short term = < 20 years.

In such streams, width/depth ratios of a channel, channel bed morphology, and relationships of a channel to the valley floor will vary significantly from reach to reach. Reach-scale variations influence the location of spawning areas and types of fish that inhabit a stream system (Grant et al. **1994;** Montgomery **1994).** Consequently, reach-scale variations are an important consideration in watershed planning.

Channel units or habitat units consist of morphological features such **as** pools, glides, riffles, rapids, stepped-pool sequences, cascades, and steps (Table 3-3). Channel units exert an important influence on local flow hydraulics and bed-sediment characteristics (Grant et al. **1990).** Channel units, therefore, provide the local habitat context for aquatic insects, fish, and other animals that inhabit stream channels.

Table 3-3. Types of channel (habitat) units. From Grant et al. (1990). FS (1993), and Beschta and Platts (1986).

Type	Morphology	Hydraulic characteristics	Ecoloaical function
Pool	Deepest, lowest gradient unit; depth varies within unit: may have asymmetrical cross-section; may accumulate fine bed material at low flows.	Slow, tranquil, sub-critical flow without hydraulic jumps during low flow; scour, turbulence, and energy dissipation during high flow.	Fish rearing; invertebrate production.
Glide (run)	Intermediate, uniform depth; symmetrical cross-section; gravel or cobble-bedded.	Tranquil subcritical flow generally without hydraulic jumps.	Fish rearing; invertebrate production.
Riffle	Shallow depth; gravel or cobble bedded.	Tranquil, generally subcritical flow with small hydraulic jumps over boulders or cobbles.	Invertebrate production: salmon and trout spawning; steelhead rearing; may be winter cover for salmon and trout; aeration.
Rapid	Shallow depth; often have transverse ribs of emergent boulders and pocket pools; common emergent boulders.	Between 15%–50% of area in supercritical flow (jumps, standing waves) at low flow.	Aeration; summer cover for salmonids.
Cascade	Shallow depth; steeper overall than rapid; consists of a series of short steps over boulders or bedrock ledges; common emergent boulders.	Greater than 50% of area in supercritical flow at low flow.	Aeration; may be migration barrier (if large).
Step	Isolated small falls, 1–2 m high and less than one channel width in length over boulders, bedrock or large woody debris; common emergent boulders, bedrock or wood; steepest and shallowest units.		Aeration; may be migration barrier (if large).

The relative abundance of different channel unit types, such as pools or cascades, varies from reach to reach in response to variation in controls such as bedrock type, reach gradient, mass movement features, sediment size, and position in the channel network. Steep reaches, associated with resistant bedrock types or with coarse substrate deposits that intersect the channel, are dominated by cascades, rapids, or steps with limited pool, glide, and riffle area (Grant et al. 1990). Overall, these channel units combine to form a step-pool channel morphology in steep reaches. The steep channel units are associated with boulder-sized bed material. Biological processes also play a role by creating steps and pools adjacent to accumulations of large woody debris (Beschta and Platts 1986). Although formed of boulder-size material that exceeds the competence of most high-flow events (i.e., mean annual floods), stepped-bed channels in steep mountain streams of the Pacific Northwest are not residual features but are in equilibrium with the modern hydrologic regime (Grant et al. 1990). The channel units are reworked by flow events with recurrence intervals of 25–50 years. In less steep reaches, the cascades, rapids and steps are less frequent; bed material is dominated by cobbles and gravels rather than boulders; and the abundance of pools, riffles and glides is higher. Active bedload, transported during frequent high flows (mean annual flood), accumulates in bars that are positive relief features on the channel bed. These bars result in pool-riffle channel morphology that is expressed at moderate to low flows. Pools are located at points of scour during high flow events, and riffles are formed by bar fronts (Lisle 1982; Beschta and Platts 1986; Wohl et al. 1993).

Human modification of flow and sediment regimes can modify the abundance and character of channel units. Human impacts resulting in net aggradation, for example, tend to reduce pool area and depth (Lisle 1982; Beschta and Platts 1986). Human impacts that decrease woody debris input to the channel can have the same effect, as can flood-induced aggradation (Lisle 1982). Human impacts that result in net degradation may also reduce pool area if bedload is depleted and bedrock is exposed in the channel bed.

3.6 Hydrology

The flow in streams and rivers represents the integration of the climate, topography, geology, geomorphology, and vegetative characteristics of a watershed. Precipitation may be intercepted by the vegetation and subsequently evaporate, or it may reach the ground either directly or as throughfall. Water reaching the ground either evaporates, infiltrates into the soil, or flows overland until it reaches the stream or an area where infiltration is possible. Water that infiltrates the soil may be taken up by plants and transpired back into the atmosphere, remain in the soil as stored moisture, percolate through the soil into deep aquifers, or enter streams via subsurface flow. Each of these processes affects the amount and timing of streamflow.

Land-use disrupts natural hydrologic processes, altering the amount of evaporation, transpiration, and runoff, the routing of water through the system, and the temporal patterns of streamflow. Regional differences in the hydrologic cycle can affect the response of a watershed to human disturbance. Consequently, an understanding of basic hydrologic processes is critical to understanding how land-use practices influence streamflow and how these effects vary across the landscape. This section provides a brief overview of hydrologic processes that occur in a watershed, with emphasis on those processes that may be substantially modified by human disturbance. A thorough review of hydrologic processes can be found in Swanston (1991).

3.6.1 Precipitation

The amount, form, and timing of precipitation differs dramatically across the Pacific Northwest, with the primary controlling factors being latitude, elevation, and proximity to the ocean and mountain ranges (Jackson 1993). Moisture-laden air generated over the Pacific Ocean is uplifted and cooled as it approaches mountainous regions, causing water to condense and fall as precipitation. After the air mass passes over these mountains it warms again, increasing its capacity to hold moisture. Thus, areas on the east slope of mountain ranges receive less rainfall than western slopes of comparable elevation, the so-called "rain shadow" effect. Convection storms (i.e., storms generated by heating and upward expansion of air masses near the earth's surface) may also be a significant source of precipitation during the spring and summer months in mountainous regions and continental climates east of the Cascade and Sierra Nevada crests. These storms tend to be localized events of high intensity and relatively short duration.

Three general precipitation systems are in the Pacific Northwest: rain-dominated, transient-snow, and snow-dominated systems (Table 3-4). Rain-dominated systems include coastal mountains, low-land valleys, and lower elevations of the Cascade and Sierra Nevada ranges, characterized by moderate to high precipitation that falls primarily as rain from late fall to early spring. In some coastal regions and lower elevations of the western Cascades, fog drip from forest canopies may also constitute a significant part of the total precipitation (Oberlander 1956; Azevedo and Morgan 1974; Harr 1982). The transient-snow zone includes mid-elevation areas of the Cascades, northern Sierra Nevada, and Olympic

Table 3-4. Precipitation patterns for selected ecoregions in the range of anadromous Pacific salmonids. Data from Omernik and Gallant (1986).

Ecoregion	Mean annual precipitation cm (inches)	Dominant form	Season of greatest precipitation
Coast Range	140 – 318 (55 – 125)	Rain.	Mid fall – early spring.
Puget Lowlands	89 – 127 (35 – 50)	Rain.	Mid fall – early spring.
Willamette Valley	89 – 114 (35 – 45)	Rain.	Mid fall – early spring.
Central CA Valley	38 – 64 (15 – 25)	Rain.	Winter.
Southern & Central CA Plains & Hills	51 – 102 (20 – 40)	Rain.	Winter.
Cascades	127 – 254 (50 – 100)	Rain (low elevation); snow (high elevation).	Mid fall – early spring.
Sierra Nevada	46 – 216 (18 – 85)	Rain (low elevation); snow (high elevation).	Mid fall – early spring.
Eastern Cascades Slopes & Foothills	30 – 64 (12 – 25)	Snow.	Mid fall – early spring.
Columbia Basin	23 – 64 (9 – 25)	Rainknow.	Fairly uniform. Fall – spring.
Blue Mountains	25 – 102 (10 – 40)	Snow.	Late fall – early spring; greater than 10% summer convective storms.
Snake River Basin/ High Desert	20 – 64 (8 – 25)	Rainknow.	Fairly uniform with slight peaks in fall and spring.
Northern Rockies	51 – 152 (20 – 60)	Snow.	Fall – spring.

Mountains that also receive most of their precipitation in the late-fall to early spring, as both rain and snow (Swanston 1991). Hydrologically, this transient zone is particularly important during rain-on-snow events. When warm, moist air-masses pass over snowpack, condensation of water on the snow surface occurs, releasing large amounts of latent energy during the phase change of water from vapor to liquid. A small amount of condensation can facilitate the rapid melting of substantial volumes of snow, which combined with runoff from rainfall can produce large floods. Snow-dominated systems include those that receive precipitation predominately as snow, including the higher elevations of the Cascade, Sierra Nevada, Olympic, and Rocky Mountain (and associated) ranges, as well as mid-elevation interior basins of the Columbia and Snake rivers. In the mountainous regions west of the Cascade and Sierra crests, precipitation is highly seasonal with most falling from fall through spring.

East of the Cascades and Sierra Nevada, the strong seasonal signature of precipitation diminishes and precipitation is spread more evenly throughout the year, particularly where spring and summer convective storms contribute substantially to the total annual precipitation. In high elevation areas of eastern Washington, the Cascades, and the Rocky Mountains, rime and hoar-frost formation may also contribute significantly to the overall water balance of a watershed (Berndt and Fowler 1969; *Gary* 1972; Hindman et al. 1983).

3.6.2 Evapotranspiration

Evapotranspiration losses include those water losses from interception by the canopy and subsequent evaporation, evaporation of water that reaches the soil, and water that enters the soil and is subsequently taken up by plants and transpired back into the atmosphere. The amount of water lost through these processes depends on vegetation type, season, and the nature of the precipitation event, including the intensity, duration, and form of the precipitation, as well as climatological conditions during the event (e.g., temperature, humidity, wind speed).

Interception Losses

Dense coniferous canopies have greater interception storage capacity than those of sparse coniferous forests, deciduous forests, shrublands, or grasslands (Wisler and Brater 1959; Zinke 1967). Rothacher (1963) reported interception and evaporation losses of nearly 100% during low-intensity rainfall events (< 0.13 cm) compared with losses of only 5%–12% during high-intensity events (> 5 cm) in an old-growth Douglas-fir forest in western Oregon. Annual interception losses for woodland-chaparral vegetation in central California ranged from 5%–8%, with seasonal losses of 4% during the winter and 14% during the spring and summer when vegetation was in full foliage (Hamilton and Rowe 1949).

Interception by coniferous canopies during snowfall can also be substantial. Snow may be temporarily stored in the canopy and then delivered to the snowpack during the storm as branches become heavily laden or following the storm by melt or wind action. Satterlund and Haupt (1970) found that 80% of the snow held in the canopy of a forest in Idaho subsequently reached the ground. Only 5% of the total snowfall was lost to interception and subsequent evaporation.

Evaporation Losses

Evaporation directly from the soil or vegetation depends on solar radiation, wind, and vapor pressure gradients between the air and the wetted soil or leaf surface. Vapor pressure gradient in the air is a function of both temperature and humidity. The temperature required for evaporation increases with increasing humidity. Under dense forest canopy, evaporation from wetted soils occurs slowly because of the high degree of shading, low temperatures, relatively high humidity, and low wind speeds typically found in these environments. More open forests allow for greater radiation and higher wind speeds that help remove water vapor from the air-soil interface, maintaining a higher vapor pressure gradient. For soil surfaces exposed to direct solar radiation, evaporation may dry soil more rapidly than transpiration because of high surface temperatures and low humidity (Satterlund and Adams 1992).

Transpiration Losses

Transpiration is the passage of water vapor from living plant tissues into the atmosphere through pores or "stomates." Transpiration rates vary based on a number of plant characteristics, including leaf surface-area, stomatal characteristics, and depth of roots; they also are affected by whether the plants are annual or perennial, and deciduous or coniferous. Coniferous forests generally have the highest leaf surface-area and thereby have the greatest potential for transpiration losses, followed in descending order by deciduous trees, shrubs, grasslands, and desert shrubs. Trees and shrubs with deeper roots can extract moisture from greater depths than grasses and forbes. Coniferous trees in xeric conditions east of the Cascades and Sierra Nevada may have large tap roots that penetrate deep into the soil, allowing moisture to be extracted even during dry periods.

Transpiration rates also depend on climatic conditions including temperature, humidity, and wind speed. In general, transpiration rates increase with increasing temperature; however, stomates will close in response to excessively high or low temperatures, increasing resistance to moisture loss. High humidity reduces the vapor pressure gradient between the plant leaf and the atmosphere, thereby reducing transpiration losses. Winds transport evaporated water vapor away from the leaf surface, thereby maintaining a higher vapor pressure gradient and increasing transpiration.

Soil conditions also influence how much water is available for transpiration. Loam soils tend to have higher water-storage capacity than sandy soils. Similarly, deep soils hold more water than shallow soils. As soil moisture is depleted, the resistance to further uptake by plants increases, and water is supplied to plants at a slower rate (Satterlund and Adams 1992). Insufficient moisture causes closure of leaf stomates, which reduces transpiration losses. Consequently, when soils are moist, transpiration

approaches maximum values; when soils are dry, transpiration is substantially less.

Total Evapotranspiration

Estimates of total evapotranspiration losses (interception + evaporation + transpiration) for a number of vegetation communities in the Pacific Northwest indicate that total losses are generally highest for coniferous forest types and slightly lower for chaparral and woodland communities (Table **3-5**); however, losses from chaparral, woodland, and semi-arid communities represent a greater percentage of total annual precipitation. This is significant in eastside systems, in part because a substantial amount of precipitation occurs during spring and fall periods when temperatures are warm and evaporation and transpiration rates are high. In contrast, precipitation in the Coast Range and western Cascades generally falls during winter, when transpiration losses are relatively low because of low solar radiation, high humidity, and cool temperatures. These differences between hydrologic processes in eastside versus westside systems are important in determining the potential effects of land-use practices; they are discussed in greater detail in Section **6.1.**

Table 3-5. Estimated precipitation and evapotranspiration for western vegetation communities.

Vegetation community	Precipitation		Evapotranspiration	
	(cm)	(inches)	(cm)	(inches)
Forest				
Lodgepole pine	51–114	20–45	48	19
Engelmann spruce-fir	51–114	20–45	38	15
White pine-larch-fir	64–152	25–60	56	22
Mixed conifer	38–178	15–70	56	22
True fir	51–254	20–100	61	24
Aspen	51–114	20–45	58	23
Pacific Douglas-fir hemlock-redwood	51–254	20–100	76	30
Interior ponderosa pine	51–76	20–30	43	17
Interior Douglas-fir	51–89	20–35	53	21
Chaparral and Woodland				
Southern California chaparral	25–102	10–40	51	20
California woodland-grass	25–102	10–40	46	18
Arizona chaparral	25–51	10–20	43	17
Pinyon-juniper	25–51	10–20	38	15
Semi-arid grass and shrub	13–51	5–20	28	11
Alpine	64–203	25–80	51	20

3.6.3 Infiltration, Subsurface Flow, and Overland Flow

The amount of water that infiltrates into the soil depends on the physical structure of the soil and antecedent moisture conditions. Sandy and gravelly soils derived from colluvium, alluvium, glacial tills, or soils that are rich in organic matter tend to be highly porous and allow rapid infiltration (Swanston 1991). Soils derived from finer-grained materials, including marine and lacustrine materials, or from weathered siltstones, sandstones, and volcanic rocks are less permeable and have lower infiltration capacities. During a given storm event, infiltration capacity decreases through time as soil pores are filled with water (Bedient and Huber 1992). If rainfall intensity (or snowmelt) exceeds infiltration capacities, overland flow occurs. Consequently, the likelihood of overland flow increases with storm intensity and duration.

In forested watersheds, most precipitation reaching the forest floor infiltrates into the soil (Satterlund and Adams 1992). Surface soils in old-growth forests areas typically have high organic content and porosity. Consequently, infiltration capacities are high, and overland flow is uncommon except in areas where soil structure has been modified through human activity or natural disturbance. The majority of water that falls on a forested landscape thereby enters streams via downslope subsurface flow. As a result, time of maximum streamflow usually lags behind peak rainfall (Swanston 1991).

In arid and semi-arid systems as well as in deforested lands, vegetation and organic litter are less abundant, and the routing of water once it reaches the soil differs. In areas where the soil surface is exposed, the impact of raindrops can detach and mobilize fine sediments (splash erosion), which settle into soil interstices, creating an impervious surface layer (Wisler and Brater 1959; Heady and Child 1994). As a result of this "rain compaction," a significant proportion of rainfall or snowmelt runs off overland to the stream. Thus, in contrast to forested watersheds, precipitation events in arid and semi-arid systems cause rapid increases in streamflow. This may be particularly evident when soils are further compacted through land-use activities.

3.6.4 Stream Hydrology

Differences in precipitation patterns, evapotranspiration rates, and infiltration processes lead to marked regional differences in hydrologic regimes of streams. In addition, the size of the drainage basin significantly influences the characteristics of streamflow at a particular point downstream. As a general rule, small headwater streams are more hydrologically dynamic than larger streams because runoff occurs more rapidly over steeper areas and because high intensity events are more common in small areas. In the discussion below, we generalize about hydrologic patterns in lower order streams.

Regional Patterns

In the Coast Range, western Cascades, Puget Lowlands, and the Willamette Valley, frequent and heavy precipitation from November to March leads to a highly variable stream hydrograph with multiple peaks that closely correspond to precipitation (Swanston 1991). In the early part of the rainy season, soil moisture is typically low, and a large fraction of rainwater functions to replenish depleted soil moisture. In addition, evapotranspiration rates decrease during the winter as temperatures drop. Consequently, precipitation events of similar intensity will result in higher peak flows in the winter, when soils are more fully saturated and transpiration demands are low, than in the fall. Streamflows are lowest during the summer when precipitation is low, evapotranspiration demands are high, and soil moisture is depleted.

In the transient-snow zone of the mid-elevations of the Cascades and northern Sierra Nevada, soils become saturated as rainfall increases in the fall. During the winter, a combination of rain and snow events occur. During rainfall events, water tends to run off quickly to the stream channel because soil moisture is high and evapotranspiration is low. Consequently, increases in streamflow tend to coincide with rainfall. Precipitation that falls as snow is stored above ground for varying lengths of time, but it generally melts within a few weeks of falling (Swanston 1991). Thus, increases in streamflow from melting snow will occur days, or even, weeks after the peak snowfall. Some of the more notable high-flow events occur when substantial snowfall is followed by high-intensity rains. These "rain-on-snow" events can release large volumes of water over short time periods.

In snow-dominated systems—the high Cascades, Sierra Nevada, Blue Mountains and northern Rocky Mountains—moisture from precipitation is stored in snowpack through much of the winter and released when temperatures warm in the late spring. Stream hydrographs are thus characterized by low winter flows followed by rapid increases during the spring snowmelt period. As snowpack diminishes, streamflow recedes and summer flows during the dry summer months typically are low, although minor peaks may result from intense convection storms. In the fall, rainstorms of moderate intensity can cause additional peaks in flow (Swanston 1991). Runoff from these events occurs most rapidly in high-elevation areas where soils are shallow and composed

of fast-draining colluvial deposits and where transpiration demands are low because of sparse vegetation.

Arid and semi-arid regions east of the Cascades and Sierra Nevada tend to have high numbers of large ephemeral and intermittent stream channels. In part, this is because the timing of precipitation can coincide with periods of relatively high solar radiation in the spring, summer, and fall, unlike west of the Cascades where most precipitation falls during cold, cloudy periods. Much of the precipitation that falls in the warmer months is either rapidly evaporated from the ground or forest canopy or transpired by vegetation. In high intensity events, sudden increases in streamflow can occur where soils are relatively impervious and water is routed rapidly to the stream channel. Those streams that flow year round are generally fed by snowmelt from higher elevations or by ground-water discharge from aquifers recharged during periods of high precipitation.

Floods

Large, infrequent floods play an important role in shaping stream channels through the erosion, transport, and deposition of bed materials. Floods with recurrence intervals of 100 years or more can result in major channel changes, and several decades may be required to re-establish an equilibrium approaching preflood conditions, Some features produced by large floods may last longer than the recurrence interval of the event (Anderson and Culver 1977), implying that large floods may be responsible for specific aspects of valley-floor formation rather than simply acting as disturbance events.

In December 1964, a rain-on-snow storm produced floods with a recurrence interval exceeding 100 years over much of northern California and Oregon. Studies conducted after this event provide information on the geomorphic effects of large floods and on the time needed to achieve a new dynamic equilibrium following such an event. The storm caused numerous debris slides and debris avalanches on slopes, and the resulting flood caused channel erosion and destruction of streamside vegetation (Lisle 1982; Sarna-Wojcicki et al. 1983). Hillslope and valley-bottom erosion put large amounts of sediment into the channel of the Van Duzen River, equal to seventeen times the mean annual sediment input into the channel system (Kelsey 1980). The result **was** a prolonged period of channel aggradation (five to fifteen years), followed by a period of degradation that was not complete in some reaches after twenty years. Erosion and the increased sediment load changed channel morphology, increasing channel width and decreasing channel depth, pool depth, and roughness (Lisle 1982). Floods of magnitude comparable to the 1964 floods occurred throughout the Willamette Valley and in southwestern Washington in February of 1996, resulting in dramatic restructuring of many stream and river channels.

The effectiveness of large floods to shape channel morphology may vary depending on stream size and position in the drainage network as well as on land cover (or recent changes in land cover such as logging). In steep mountain streams, only large, infrequent floods significantly modify valley-floor landforms. In lower gradient alluvial reaches, smaller, more frequent events and ongoing processes modify the valley floor (Grant et al. 1994). Floods also deposit sediments onto the surrounding floodplain, transport and rearrange large woody debris within the channel, clean and scour gravels in streams, recharge floodplain aquifers, and disperse propagules of riparian vegetation.

Droughts

Below-average precipitation and runoff can have significant effects on streams and watersheds. The recent drought in the Pacific Northwest has focused much attention on the health of forest ecosystems east of the Cascade Crest (Quigley 1992). Substantial die-off of forest vegetation has resulted from the synergistic effects of fire suppression and forest practices, which have led to changes in species composition of terrestrial vegetation (see Section 3.3). Drought conditions have weakened trees, making them more vulnerable to infestation by insects or disease. The influence of drought on watershed processes is not well documented; however, it is likely that droughts affect the input of nutrients, allochthonous materials, and large woody debris to stream channels. Within the stream channel, low flows can constrict the available habitat and allow water temperatures to warm, stressing fish or creating thermal barriers that block migration. A potential benefit of drought is that it provides the opportunity for establishment of riparian vegetation within the active stream channel, which in turn can stabilize channel features, dissipate hydraulic energy, and collect sediment when flows rise again (Blau 1995). Tree-ring records from eastside forests indicate that a number of significant droughts lasting from 5 to 20 years have occurred during the past 300 years (Agee 1994).

3.7 Thermal Energy Transfer

Because most aquatic organisms are ectothermic, water temperature plays an important role in regulating biological and ecological processes in aquatic systems. Temperature directly and indirectly affects physiology, development, and behavior of

salmonids, as well as mediates competitive interactions, predator-prey relationships, and the incidence of parasitism and disease (see Chapter 4). Land-use practices can significantly change seasonal and diel temperature regimes in streams, primarily through the alteration of forest and riparian canopy but also through irrigation, impoundments, heated industrial effluents, and thermal power plants. In this section, we review the dominant energy transfer processes that are responsible for the heating and cooling of streams, rivers, and lakes to provide the basis for evaluating the effects of land-use practices on salmonid habitat. The role of riparian vegetation in controlling these processes is emphasized.

3.7.1 Heaf Exchange in Streams

Heat energy is transferred to and from streams and rivers by six processes: short-wave radiation (primarily direct solar), long-wave radiation, convective mixing with the air, evaporation, conduction with the stream bed, and advective mixing with inflow from groundwater or tributary streams (Beschta et al. 1987; Sullivan et al. 1990). These processes occur in all streams, but the importance of each process on stream temperatures varies with location and season (Sullivan et al. 1990).

Direct solar radiation is generally the dominant source of energy input to streams and rivers. The amount of solar radiation that reaches and is absorbed by streams and rivers is influenced by season, latitude, topography, orientation of the watershed, local climate, and riparian vegetation. Season and latitude together determine the amount of daylight and the solar angle, both of which affect the amount of energy absorbed by streams (Brown 1980). In mountain or canyon regions, topography may provide substantial shade to streams, particularly at times of the year when the sun is low in the sky and in north-facing drainages. Local climate, and particularly cloud cover, significantly influences how much solar radiation reaches the stream channel. The amount and type of riparian vegetation play dominant roles in regulating incoming solar radiation in smaller streams (Brown 1980; Beschta et al. 1987; Caldwell et al. 1991). The percentage of total solar radiation that reaches the stream surfaces in forested reaches may vary from less than 16% under dense coniferous canopies found in old-growth stands of the Coast Range and western Cascades (Summers 1983) to 28% in old-growth forests east of the Cascades (Anderson et al. 1992, 1993). In alpine, arid, and semi-arid ecosystems, the degree of shading may be less. Deciduous vegetation can provide significant shading during the spring and summer months, but it has minimal effect after leaf drop in the fall. The influence of riparian vegetation on radiation inputs diminishes in a downstream direction. As streams become larger and wider, riparian vegetation shades a progressively smaller proportion of the water surface (Beschta et al. 1987).

Long-wave radiation back into the atmosphere plays a relatively minor role in the overall energy budget of a stream. Long-wave radiation loss is determined primarily by the temperature differential between water and air, with greater exchange occurring when the difference between the air and water temperatures is greatest. Riparian vegetation reduces long-wave radiation through its effect on microclimate within the riparian zone. Temperatures in the riparian zone tend to be cooler during the day and warmer at night than those above the forest canopy; this dampening of diel temperature fluctuations moderates long-wave radiative gains and losses.

Convective and evaporative heat transfer are controlled by temperature and vapor-pressure gradients, respectively, at the air-water interface (Beschta et al. 1987). Greater convective exchange occurs when the temperature differential between air and water is highest. Similarly, evaporative losses are highest at low humidity. Wind facilitates both convective and evaporative losses by displacing air near the air-water interface as it approaches thermal equilibrium with the water and as it becomes more saturated through evaporation. Riparian vegetation modifies convective and evaporative heat-exchange losses by creating a microclimate of relatively high humidity, moderate temperatures, and low wind speed compared with surrounding uplands. These microclimate conditions tend to reduce both convective and evaporative energy exchange by minimizing temperature and vapor-pressure gradients.

Conductive transfer of heat generally represents a minor component of a stream heat budget. The amount of heat transferred depends on the nature of the substrate, with bedrock substrates being more efficient in conducting heat than gravel beds (Beschta et al. 1987). Brown (1980) estimates that heat flow into bedrock stream beds may be as high as 15%–20% of the incident heat. Heat that is transferred to the streambed during the daylight hours serves to heat streams during periods of darkness, thereby dampening diel fluctuations. In shallow, clear streams, without shade from riparian vegetation, solar energy may penetrate through the water column and heat the substrate directly.

The role of advection depends on the volume of groundwater or tributary inputs relative to the total stream discharge; consequently, the importance of advection tends to diminish in a downstream direction. Nevertheless, even when groundwater inputs to streams are small, they may provide thermal heterogeneity that is biologically important (see Sections 5.2.1 and 5.2.3). In addition, certain

regions east of the Cascade Range (e.g., the Deschutes Basin) are underlain with porous basaltic formations that absorb large amounts of water during periods of high runoff and release it later in the year. These groundwater inputs can significantly moderate streamflow and temperature regimes in both summer and winter.

As subsurface flow moves laterally and downward towards stream beds, water temperatures equilibrate with those in the subsurface soil layers (Beschta et al. 1987); consequently, the temperature of water that enters streams from groundwater flow depends on ambient conditions in the soil environment. Surface-soil temperatures follow seasonal air temperature patterns with a time lag that increases with increasing depth (Meisner 1990). Seasonal fluctuations are greatest at the surface and decrease with depth down to the "neutral zone," generally about 16–18 m below the surface, where temperatures remain constant throughout the year (Meisner 1990). If the groundwater flow originates below the neutral zone, then groundwater temperatures will remain constant; if it originates above the neutral zone, then groundwater temperatures will exhibit seasonal variation (Meisner 1990). Melting snow infiltrates into the soil at temperatures approaching 0°C in snow-dominated systems (Beschta et al. 1987).

3.7.2 Stream Temperature Regulation

All of the above processes interact to produce the temperature regimes observed in streams and rivers; however, the relative importance of each process differs among locations. In small- to intermediate-sized streams of forested regions, incoming solar radiation represents the dominant form of energy input to streams during summer, with convection, conduction, evaporation, and advection playing relatively minor roles (Brown 1980; Beschta et al. 1987; Sullivan et al. 1990). Groundwater inputs may be important in small streams where they constitute a large percentage of the overall discharge, particularly during periods of the year when flows are low. Downstream, where flow increases, the effects of riparian shading and advective mixing generally diminish, and the importance of evaporative heat-loss increases.

Channel characteristics may also significantly affect heat-exchange processes. The amount of heat that is gained or lost and the rate at which exchange takes place depend on the surface area of the stream or river. Wide, shallow streams exhibit greater radiative, convective, and evaporative exchange and, consequently, heat and cool more rapidly than deep, narrow streams. Similarly, the rate of energy exchange is affected by seasonal changes in stream discharge, which alter surface-to-volume ratios and determine the relative importance of groundwater inputs. In most streams in the Pacific Northwest, groundwater inputs are critical to cool streams during warm summer months. Regional differences in stream temperatures result from differences in climatic factors (e.g., humidity, air temperature). Streams in the Coast Range and western Cascades are moderated by the maritime climate and undergo smaller seasonal temperature fluctuations than those in the continental climates east of the Cascades. Elevation also influences stream temperatures, primarily because of elevational gradients in air temperatures that lead to greater convectional heating (Beschta et al. 1995). Finally, high turbidity in streams and rivers substantially increases the absorption of high-energy, shortwave radiation (Wetzel 1983) and thereby can affect stream heating.

3.7.3 Lakes and Reservoirs

Lakes and reservoirs are heated primarily by incoming solar radiation, although some heat is transferred by convection, conduction (in shallow waters), and evaporation (Wetzel 1983). In clear water, over one-half of the incoming solar radiation is absorbed in the upper two meters of water, and more may be absorbed in waters with high turbidity. In temperate lakes, incoming solar radiation exceeds outgoing long-wave radiation during the summertime, and water at the surface is gradually warmed. Because warm water is less dense than cold water, it tends to remain near the surface and is resistant to mixing by the wind. **As** a result, thermal stratification can occur with a warm and relatively well-mixed "epilimnion" overlying a cooler "hypolimnion." Between these two layers is a transition zone, or "metalimnion," where temperatures rapidly decrease with increasing depth. During the fall as solar radiation decreases, temperatures in the surface layers cool, and the mixing of epilimnetic and hypolimnetic waters occurs as they reach comparable temperature and density. During the winter, lake waters tend to remain mixed except where temperatures are sufficiently cold to cause freezing. Because water reaches maximum density at 4°C, lakes that freeze are colder near the surface and warmer near the bottom (inverse stratification). When lakes become ice-free in the spring, density is relatively uniform, and mixing of the water column will occur again provided there is sufficient wind at the surface.

The above pattern is characteristic of deeper lakes in the Pacific Northwest. In shallower lakes and ponds, lakes may turn over many times each year, whenever high wind conditions occur. Such systems are usually poor habitat for salmonids because they warm throughout the water column.

Thermal structure plays an important role in determining the distribution and production of aquatic organisms within a lake or reservoir. Stratification of lakes may restrict the habitats of fishes and other aquatic organisms. Moreover, during the spring and fall mixing periods, the circulation (turnover) of water brings nutrient-rich waters to the surface and stimulates production of phytoplankton and zooplankton. Release of either epilimnetic or hypolimnetic water from stratified reservoirs can markedly influence downstream temperature regimes in ways that may have adverse consequences for fish.

3.8 Nutrient Cycling/Solute Transport

Water is the major agent for the flux of dissolved and particulate matter across the landscape, integrating processes of chemical delivery in precipitation, geologic weathering, erosion, chemical exchange, physical adsorption and absorption, transport and retention in surface waters, and biotic uptake and release. At any point within a landscape or catchment, concentrations of nutrients or suspended material result from many abiotic and biotic processes.

Geology, climate, and biological processes across a landscape determine patterns of nutrient cycling. The primary determinant of the chemistry of most surface waters is the composition and age of the parent geology. The major rock types—igneous, sedimentary, and metamorphic—have characteristic compositions of major cations and anions, as well as minor chemical constituents that serve as nutrients (e.g., nitrogen, phosphorus) for biota. The high temperatures and pressure under which igneous and metamorphic rocks are formed alters the chemical composition by volatilizing elements and compounds that are released as gases (e.g., nitrogen, inorganic carbon) to the atmosphere. Sedimentary rocks contain minerals that have been weathered from other sources previously, and they may contain greater amounts of biologically derived material because of the less harsh conditions of their formation. Geochemistry of the parent material governs rates of dissolution or weathering and, thus, influences concentrations of dissolved chemicals in surface waters.

Climate strongly influences general surface-water chemistry and nutrient concentrations through two major processes—direct input of chemicals through precipitation and influence on hydrology (Gibbs 1970). The atmosphere is a major source of elements and compounds. Weather patterns affect the available source areas for water and chemicals in the atmosphere and subsequent precipitation that falls on land. Both natural and anthropogenic sources may create distinctive chemical signatures in precipitation. Climate also determines the general hydrologic regime and establishes physical conditions that influence evaporative losses of water. Arid areas typically exhibit high concentrations of dissolved ions because of high rates of evaporation and subsequent concentration of chemicals in solution. The hydrologic regime is a function of climate and geographic features of the landscape, and it is a major determinant of weathering rates, dilution, and timing of nutrient transport. Patterns in runoff may be mirrored by differences in surface water chemistry. The flashy flow-regimes of rain- and rain-on-snow dominated systems create a similar episodic pattern in nutrient transport, while the more steady flow regimes of snow-dominated systems produce more predictable nutrient transport patterns.

The biota of terrestrial, riparian, and aquatic ecosystems strongly influence the cycling of major nutrients and associated chemical parameters (Likens et al. 1977; Meyer et al. 1988) through such processes as photosynthesis, respiration, food consumption, migration, litter fall, and physical retention. Surface waters are exposed to various sources of inputs, sites of biological uptake, and surfaces for physical exchange (Gregory et al. 1991). Stream substrates serve as sites for colonization and attachment by aquatic organisms ranging from microbes to vertebrates. Many aquatic organisms have distinct substrate relationships; therefore, the composition of the stream bed can directly influence nutrient cycling. Organic substrates, such as leaves and wood, create important sources for microbial colonization and subsequent nutrient cycling (Aumen et al. 1985a, 1985b; Meyer et al. 1988). These organic substrates also serve as sources of dissolved organic carbon for microbial activity or transport into the water column (Dahm 1981). Woody debris in particular plays a critical role as a food resource, substrate, site of physical exchange, site for biological uptake, and roughness element that reduces water velocity and increases retention (Harmon et al. 1986). Land-use practices typically alter the organic substrates of stream channels, and thereby influence water quality.

Vegetated floodplains along streams and rivers as well as mudflats and vegetation beds in estuaries create a mosaic of geomorphic surfaces and riparian plant communities (Fonda 1974; Gregory et al. 1991; Bayley and Li 1992). Floodplains influence the delivery and transport of material by 1) delivering stored material during high flows, 2) retaining material in transport from the main channel, 3) providing a matrix of sediment for subsurface flow, and 4) reducing velocities of water and increasing the potential for retention. Elimination of floodplains greatly reduces the assimilative and storage capacity of a stream system and is one of the major forms of anthropogenic alteration of nutrient cycling in lotic

ecosystems (Smith et al. 1987; Junk et al. 1989; Sparks et al. 1990). Side channels on floodplains and in estuaries are habitats with extensive contact with the water column and lower velocities than the main channel; consequently, these lateral habitats typically exhibit high rates of nutrient uptake and biological productivity (Cooper 1990).

Streamside forests, estuarine vegetation beds (tidal marshes), and other plant communities create a filter through which nutrients in solution must pass before entering surface waters (Pionke et al. 1988; Gregory et al. 1991). Retention of nutrients in groundwater is a critical component of nutrient cycling within a basin (Simmons et al. 1992). Commonly, these vegetative corridors remove 60%–90% of the nitrogen and phosphorus in transport (Lowrance et al. 1984; Peterjohn and Correll 1984; Lowrance 1992). Modification of riparian forest structure can substantially change long-term patterns of nutrient cycling within a catchment (Pinay et al. 1992).

One of the most overlooked components of a stream and its valley is the hyporheic zone, the area of flow beneath the surface of the stream bed (Stanford and Ward 1988; Bencala 1993). In alluvial valleys, the hyporheic zones may extend several meters below the channel bed, as well as a kilometer or more laterally. Recent research indicates the hyporheic zone plays important roles in nutrient cycling, temperature modification, dissolved oxygen microbial processes, meiofaunal communities and refugia for a wide range of organisms (Pinay and Decamps 1988; Stanford and Ward 1988; Triska et al. 1990; Valett et al. 1990; Hendricks and White 1991). In many streams, **as** much as 30%–60% of the flow occurs in the hyporheic zone and may exceed these levels in porous bed materials or during low flow conditions. The majority of nutrient uptake in streams may occur in the hyporheic zone in desert, forest, or grassland ecoregions (Duff and Triska 1990).

3.8.1 Major Chemical Species and Dissolved Nutrients

Surface waters contain a complex array of major chemical species, biologically important nutrients, and numerous trace elements and compounds. The major dissolved constituents include cations and anions that are required by living organisms but are so abundant that they rarely limit biological production. In addition, surface waters contain concentrations that they limit rates of production of plants, microbes, or consumers. The major nutrients or macronutrients are nitrogen, phosphorus, and carbon. Micronutrients are generally required in such low amounts that their availability is rarely limiting, but studies over the last several decades have demonstrated that the productivity of some systems may be limited by micronutrients and many processes are commonly limited by the availability of these chemicals. This review only covers the macronutrients.

The major cations in surface waters include the divalent cations of calcium and magnesium and the monovalent cations of sodium and potassium. In general, the order of dominance in surface waters of the world is $Ca^{++} > Mg^{++} > Na^{+} > K^{+}$, but local geology can alter their relative abundance (Gibbs 1970). These elements play critical roles in all biological systems as well as influence the reactivity and abundance of other elements. The exchange of these cations, either physically or through biological absorption, can alter the availability of hydrogen ions and thereby alter pH, which strongly influences biota and fundamental ecological processes.

The major anions in surface waters consist of the divalent anions of carbonate and sulfate and the monovalent anions of bicarbonate and chloride (Gibbs 1970). In temperate waters, the dominance of anions is ordered: $HCO_3^- > CO_3^{--} > SO_4^{--} > Cl^-$. Inorganic carbon and sulfate are biologically important in all ecosystems, and the inorganic carbon species largely determine the buffering capacity and, consequently, the pH conditions of surface waters.

Nitrogen

Nitrogen exists in solution as both inorganic forms—nitrogen gas (N_2), nitrate (NO_3^-), nitrite (NO_2^-), ammonia (NH_3), or ammonium (NH_4^+)—and organic forms (organic N). In many areas of the Pacific Northwest, surface waters commonly have extremely low concentrations of dissolved nitrogen because of the underlying volcanic parent geology, which was created under intense temperature and pressure (Thut and Haydu 1971; Sollins and McCorison 1981; Norris et al. 1991).

Biological processes largely mediate the different forms of nitrogen (Gosz 1981). Nitrogen fixation converts N_2 into NH_3 under anaerobic conditions or in specialized cells, and organisms subsequently use the ammonia to form amino acids and proteins. Organic nitrogen is metabolized to ammonium as a waste product or microbial decomposition converts organic N to ammonium through the process of ammonification. Certain microorganisms are capable of oxidizing ammonia to nitrite or nitrate. Plants and heterotrophic microorganisms can then reduce nitrate to form ammonia and subsequently proteins and amino acids. Under anoxic conditions, certain microorganisms can reduce NO_3^- to N_2. These transformations create intricately linked cycles of nitrogen, and under nitrogen-limited conditions, these links are tightly coupled. **As a** result, certain forms—such as ammonia or nitrate—are rarely

present naturally in high concentrations because they are so rapidly incorporated into other nitrogenous molecules or are modified.

Riparian areas play major roles in nitrogen cycling by providing year-round anaerobic conditions (Green and Kauffman 1989; Mulholland 1992). Rates of denitrification (and nitrogen fixation) are enhanced in the anaerobic conditions and in the high moisture and organic substrates that denitrifying bacteria require (Myrold and Tiedje 1985; Ambus and Lowrance 1991; Groffman et al. 1991). Rates of denitrification in riparian soils in the Cascade Mountains of Oregon are four to six times higher than in upslope forests, and alder-dominated reaches exhibit the highest observed rates (Gregory et al. 1991). Alder is a common streamside plant and is also a nitrogen fixer; thus, alder-dominated riparian areas are potential sources of nitrogen in stream water (Tarrant and Trappe 1971). As noted above, elevated rates of denitrification may negate the contribution of alders, but it is possible for extremely high concentrations of nitrate (> 5 mg NO_3-N/l) to occur where litter inputs are high and water velocities are low (Taylor and Adamus 1986). These conditions have been observed primarily in the Coast Range where alder may extend from stream's edge to the ridgeline.

Catchments generally process nitrogen efficiently because it is such a biologically important element. A small western basin retained approximately 99% of the nitrate that entered in precipitation (Rhodes et al. 1985). Loss of nitrogen from terrestrial ecosystems is mediated by uptake in the aquatic ecosystems, particularly in nitrogen-limited ecosystems, such as the basalt-dominated Pacific Northwest (Triska et al. 1982, 1984). Studies of nitrogen uptake in streams of the Cascade Mountains indicate that approximately 90% of the nitrate or ammonium introduced into stream water is assimilated within 500–2,000 m, depending on the size of the stream (Lamberti and Gregory 1989; D'Angelo et al. 1993).

Phosphorus

Phosphorus in surface waters is largely derived from mineral sources. Inorganic phosphorus includes many compounds incorporating the phosphate ion (PO_4^{---}). Concentrations of inorganic phosphorus are low in many geologic areas, and as a result, phosphorus commonly is a limiting nutrient for primary production and microbial processes (Wetzel 1983). In much of the Pacific Northwest, however, the basaltic parent geologic material contains abundant and relatively easily weathered forms of inorganic phosphorus; thus, concentrations of phosphorus in streams and rivers commonly exceed 10 μg PO_4–$P \cdot L^{-1}$ (Fredriksen et al. 1975; Salminen and Beschta 1991; Bakke 1993).

3.8.2 Nutrient Spiraling and Retention

Nutrient cycling is often viewed as a closed system in which chemicals pass through various states and reservoirs within the ecosystem of interest. Stream ecosystems present an interesting contrast to this perspective because of their unidirectional flow from headwaters to large rivers to the ocean. The Nutrient Spiraling Concept was developed to more accurately represent the spatially dependent cycling of nutrients and the processing of organic matter in lotic ecosystems (Newbold et al. 1982; Elwood et al. 1983).

The longitudinal nature of streams and rivers strongly influences patterns of nutrient uptake. In the Nutrient Spiraling Concept, one complete cycle of a nutrient depends upon the average distance a nutrient atom moves in the water compartment (i.e., the uptake length), the average distance a nutrient atom moves in the particulate compartment, and the average distance a nutrient atom moves in the consumer compartment. The Nutrient Spiraling Concept provides a useful framework to investigate the dynamics of dissolved and particulate material in streams and rivers (Mulholland 1992). Alteration of riparian areas, stream channels, and biotic assemblages can be viewed in terms of changes in flux and uptake, the two major components of spiraling length. Efficiency of nutrient use can be quantified in terms that are relevant to the cycling of nutrients along a river valley or drainage network.

Downstream transport of dissolved or particulate material is a complex function of physical trapping, chemical exchange, and biological uptake (Minshall et al. 1983; Speaker et al. 1984). Retention of material in streams is not necessarily uniform along a reach of stream. Physical discontinuities, such as debris dams, boulders, pools, and sloughs, alter retention patterns. The ionic strength or salinity of surface water tends to increase from headwaters to large rivers, reflecting the accumulation of weathering products and material produced by terrestrial and aquatic ecosystems (Dahm et al. 1981). Spiraling length increases and retention efficiency decreases as streams become larger because of the decreased friction, increased average velocity, and lower probability of being trapped by bed material. This pattern is moderated in braided channels and at high flows as streams flow out of their banks and are slowed by the roughness of adjacent forests and floodplains (Welcome 1988; Junk et al. 1989; Sparks et al. 1990).

Different environments may alter retention patterns for dissolved and particulate matter. Areas of intense biological activity increase biotic uptake and alter patterns of retention. Simplification of stream ecosystems will tend to make longitudinal

patterns of retention more uniform and less efficient, thus lowering biological productivity.

Retention of material represents a fundamental ecological feature that integrates the supply and use of nutrients and food resources. Historically, salmon and lamprey carcasses brought significant quantities of organic matter from the Pacific Ocean into freshwater ecosystems of the Pacific Northwest (Bilby and Bisson 1992). The abundance of salmon carcasses has long been correlated with the productivity of sockeye lakes in Alaska for the subsequent year class (Donaldson 1967). Recent studies have demonstrated that as much as 30% of the nitrogen for higher trophic levels in streams in the Pacific Northwest may be derived from marine ecosystems (Bilby et al. 1996). In addition, retention of carcasses in streams has been linked to channel complexity and abundance of woody debris (Cederholm and Peterson 1985). Declines in anadromous fishes in the Pacific Northwest may signal more fundamental changes in productivity of stream ecosystems than the simple loss of stocks or species.

Disturbances can accelerate or slow the loss of nutrients and the efficiency with which terrestrial and aquatic ecosystems use them (Vitousek and Melillo 1979; Beschta 1990). Generally, such disturbances disrupt nutrient cycling over the short-term (i.e., less than a decade) (Resh et al. 1988), but as ecosystems recover, they more efficiently cycle available nutrients. Many disturbances also increase habitat complexity (Swanson et al. 1982a), thereby increasing the efficiency of retention after an initial recovery period (Bilby 1981; Aumen et al. 1990). The frequent disturbances associated with stream ecosystems make them one of the most dynamic ecosystems with respect to nutrient cycling and biotic community organization (Minshall et al. 1985; Minshall 1988). Changes in community organization and process rates in response to changes in long-term nutrient availability may not be fully exhibited for years (Stottlemyer 1987; Power et al. 1988; Peterson et al. 1994).

3.9 Roles of Riparian Vegetation

Riparian zones constitute the interface between terrestrial and aquatic ecosystems (Swanson et al. 1982b; Gregory et al. 1991), performing a number of vital functions that affect the quality of salmonid habitats as well as providing habitat for a variety of terrestrial plants and animals. While processes occurring throughout a watershed can influence aquatic habitats, the most direct linkage between terrestrial and aquatic ecosystems occurs in the riparian area adjacent to the stream channel. Consequently, the health of aquatic systems is inextricably tied to the integrity of the riparian zone (Gregory et al. 1991; Naiman et al. 1992).

Riparian vegetation provides numerous functions including shading, stabilizing streambanks, controlling sediments, contributing large woody debris and organic litter, and regulating the flux and composition of nutrients (FEMAT 1993; O'Laughlin and Belt 1994; Cederholm 1994). Riparian-aquatic interactions are now recognized by scientists as so important that riparian buffers have been established as a central element of forest practices rules and watershed restoration efforts. Authors in several recent publications have advocated a functional approach to riparian management, attempting to identify "zones of influence" for critical riparian processes (McDade et al. 1990; FEMAT 1993; O'Laughlin and Belt 1994). These approaches recognize that the influence of riparian vegetation on stream ecosystems generally diminishes with increasing distance from the stream channel. In this section, we review principal functions of riparian vegetation and summarize the available literature on zones of riparian influence. Riparian zones of influence and effective riparian buffer widths are elaborated further in Part II, Section 14.2.3.

3.9.1 Shade

In small headwater streams, riparian vegetation moderates the amount of solar radiation that reaches the stream channel, thereby dampening seasonal and diel fluctuations in stream temperature (Beschta et al. 1987) and controlling primary productivity. The effectiveness of riparian vegetation in providing shade to the stream channel depends on local topography, channel orientation and width, forest composition, and stand age and density (Beschta et al. 1987; FEMAT 1993). Naiman et al. (1992) report that in westside forests the amount of solar radiation reaching the stream channel is approximately 1%–3% of the total incoming radiation for small streams and 10%–25% for mid-order streams. In winter, streamside vegetation provides insulation from radiative and convective heat losses (see Section 3.7.1), which helps reduce the frequency of anchor-ice formation (Murphy and Meehan 1991). Thus, riparian vegetation tends to moderate stream temperatures year round. The numerous biological and ecological consequences of elevated stream temperatures on salmonids include effects on physiology, growth and development, life history patterns, competitive and predator-prey interactions, and disease (see Section 4.3).

The FEMAT (1993) report presents a generalized curve relating cumulative effectiveness of the riparian canopy in providing shade relative to distance from the stream channel for westside forests (Figure 3-2).

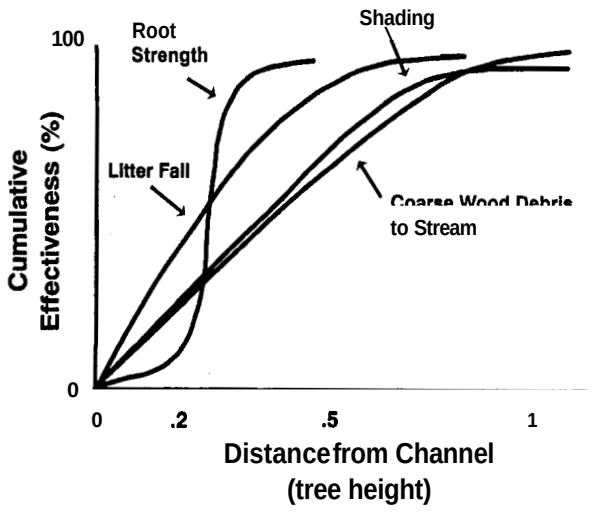


Figure 3-2. Riparian forest effect on streams as a function of buffer width. From FEMAT (1993).

They propose that for these systems close to 100% of the potential shade value (rarely complete canopy cover) can be maintained by buffer widths equal to one site-potential tree height (i.e., the potential height of a mature tree at the particular location). In the Oregon Coast Range and western Cascades, buffer zones of 100 feet or more can provide as much shade as intact old-growth forests (Brazier and Brown 1973; Steinblums et al. 1984). Similar assessments for eastside forests as well as arid and semi-arid shrublands have not been published; effective buffers widths in these systems may differ substantially.

3.9.2 Bank Stabilization

Riparian vegetation increases streambank stability and resistance to erosion via two mechanisms. First, roots from woody and herbaceous vegetation bind soil particles together, helping to maintain bank integrity during erosive high-streamflow events (Swanson et al. 1982b). Diverse assemblages of woody and herbaceous plants may be more effective in maintaining bank stability than assemblages dominated by a single species; woody roots provide strength and a coarse root network, while fine roots fill in to bind smaller particles (Elmore 1992). The root matrix promotes the formation of undercut banks, an important habitat characteristic for many salmonids (Murphy and Meehan 1991). Second, stems and branches moderate current velocity by increasing hydraulic roughness. East of the Cascades, grasses, sedges, and rushes tend to lie down during high flows, dissipating energy and protecting banks from erosion (Elmore 1992).

Riparian vegetation may also facilitate bank-building during high flow events by slowing stream velocities, which in turn helps to filter sediments and debris from suspension. This combing action helps to stabilize and rebuild streambanks, allowing the existing channel to narrow and deepen, and increasing the effectiveness of riparian vegetation in providing bank stability and shade (Elmore 1992). During overbank flows, water is slowed and fine silts are deposited in the floodplain, increasing future productivity of the riparian zone.

Vegetation immediately adjacent to the stream channel is most important in maintaining bank stability. The FEMAT (1993) report suggests that the role of roots in maintaining streambank stability is negligible at distances of greater than 0.5 tree heights from the stream channel (Figure 3-2). In wide valleys where stream channels are braided, meandering, or highly mobile, the zone of influence of root structure is substantially greater.

3.9.3 Sediment Control

The regulation of sediment flow is a major function of the riparian zone. Riparian vegetation and downed wood in the riparian zone can reduce the amount of sediment delivered from upland areas to the stream channel in several ways. By providing physical barriers, standing or downed vegetation can trap sediments moving overland during rainfall events. Riparian zones, however, are less effective in regulating channelized erosion. Most surface erosion occurs in channelized flows that may travel thousands of feet (Belt et al. 1992; O'Laughlin and Belt 1994). Thus, riparian vegetation may have little influence on sediments derived from outside of the riparian zone. Riparian vegetation may also influence sediment inputs by reducing the likelihood of mass failures along the stream channel through the stabilizing action of roots and by buffering the stream from mass wasting that initiates in upland areas, although riparian vegetation may have little effect during large, deep-seated landslides (Swanson et al. 1982b).

The zone of riparian influence for sediment regulation is difficult to define because of different ways sediment may enter the stream channel. The FEMAT (1993) review of the literature suggests that riparian zones greater than 200 feet (i.e., about one site-potential tree height) from the edge of the floodplain are probably adequate to remove most sediment from overland flow. However, O'Laughlin and Belt (1994) suggest sediment control cannot be achieved through riparian zones alone because channel erosion and mass wasting are significant sources of sedimentation in forested streams. For these events the zone of influence may extend several hundred meters from the floodplain (FEMAT 1993), depending on the soil type, slope steepness, and other factors that influence the susceptibility of hillslopes to mass wasting or channelized erosion.

3.9.4 Organic Litter

Riparian vegetation contributes significant quantities of organic litter to low- and mid-order streams. This litter constitutes an important food resource for aquatic communities (Naiman et al. 1992). The quality, quantity, and timing of litter delivered to the stream channel depends on the vegetation type (i.e., coniferous versus deciduous), stream orientation, side slope angle, stream width, and the amount of stream meander (Cummins et al. 1994). In conifer-dominated riparian zones, 40%–50%of the organic litter consists of low quality cones and wood, which may take several years to decades to be processed. In contrast, high quality material from deciduous forests may decay within a year. Although conifers have the greater standing biomass, shrub- and herb-dominated riparian assemblages provide significant input in many streams (Gregory et al. 1991). Over 80% of the deciduous inputs, primarily leaves, are delivered during a 6–8 week period in the fall (Naiman 1992), while coniferous inputs are delivered throughout the year (Cummins et al. 1994).

The extent of the riparian zone of influence for organic-litter inputs depends on geomorphology and stream size. Upland forests beyond the riparian zone can contribute litter to small streams in steep basins through direct leaf-fall and overland transport of material by water. Larger streams (3rd to 5th order) are more influenced by vegetation in the immediate riparian zone. Large lowland streams tend to have complex floodplain channels with minimal upland interactions. However, the lateral movement of unconstrained alluvial channels effectively increases the potential riparian zone of influence. In westslope forested systems, most organic material that reaches the stream originates within 0.5 tree heights from the stream channel (Figure 3-2) (FEMAT 1993). Vegetation type may also influence the riparian zone of influence because deciduous leaves may be carried greater distances by the wind than coniferous litter.

3.9.5 Large Woody Debris

Large woody debris (LWD) provides critical structure to stream channels, although full recognition of the importance of large wood in stream ecosystems has only come in the last 20 years (Swanson et al. 1976; Swanson and Lienkaemper 1978; Harmon et al. 1986). For more than 100 years, large wood was removed from stream channels in the United States to facilitate boat traffic and the floating of **logs** downstream. In addition, up until the late 1970s and early 1980s, biologists viewed large wood as an impediment to fish migration and recommended clearing woody debris from stream channels (Sedell and Luchessa 1981). Consequently, the many roles of large wood in streams, from small headwaters to large river systems, have been greatly diminished over time.

Large wood enters the stream channel through two different pathways: the steady toppling of trees as they die or are undercut by streamflow, and catastrophic inputs associated with windstorms, mass failures, and debris torrents (Bisson et al. 1987; Cummins et al. 1994). Once in the stream channel, large woody debris influences coarse sediment storage; increases habitat diversity and complexity, gravel retention for spawning habitat, and flow heterogeneity; provides long term nutrient storage and substrate for aquatic invertebrates; moderates flow disturbances; increases retention of allochthonous inputs, water, and nutrients; and provides refugia for aquatic organisms during high- and low-flow events (Bisson et al. 1987). The ability of large wood to perform these functions depends in part on the size and type of wood. In general, the larger the size of the debris, the greater its stability in the stream channel, since higher flows are needed to displace larger pieces (Bilby and Ward 1989). In addition, coniferous logs are more resistant to decay than deciduous logs and hence exhibit greater longevity in the stream channel (Cummins et al. 1994).

Although LWD performs essential functions in all streams, the relative importance of each of the processes listed above varies with stream size. In small, steep headwater streams (1st and 2nd order), large volumes of stable LWD tend to dominate hydraulic processes. Generally, woody debris is large enough to span the entire channel, resulting in a stepped longitudinal profile that facilitates the formation of plunge pools downstream of obstructions (Grant et al. 1990). This stepped profile increases the frequency and volume of pools, decreases the effective streambed gradient, and increases the retention of organic material and nutrients within the system, thus facilitating biological processing (Bisson et al. 1987). Woody debris within the channel increases velocity heterogeneity and habitat complexity by physically obstructing the streamflow, creating small pools and short riffles (Swanston 1991). Diverted currents create pools (plunge, lateral, backwater) and riffles, flush sediments, and scour streambanks to create undercut banks (Cummins et al. 1994). In sediment-poor systems, LWD retains gravels that are essential for spawning salmonids. Larger debris-dams store fine sediment and organic materials, reducing their rate of transport downstream. In addition debris-dams protect the downstream reaches from rapid changes in sediment loading, which may degrade spawning gravels, fill pools, and reduce invertebrate populations.

In mid-order streams, large woody debris functions primarily to increase channel complexity and flow heterogeneity by 1) anchoring the position of pools along the thalweg, 2) creating backwaters along the stream margin, **3)** causing lateral migration of the channel, and 4) increasing depth variability (Maser et al. 1988). Large wood deposits tend to occur along margins, or in mid-channel where physical obstructions such as gravel bars collect wood (Bisson et al. 1987). Bilby and Ward (1989) examined streams in western Washington and found a number of differences in the roles of large wood in relation to stream size. Average diameter, length, and volume of pieces of wood were generally greater in mid-order streams than in low-order streams. Large wood was important in pool formation in mid-sized streams; however, these were more likely to be debris-scour pools than plunge pools. In addition, the ability of wood to accumulate sediment diminished as streams became larger, a result of increased stream power. Distributions of organisms associated with woody debris, including various salmonids, changed relative to the changes in woody debris distribution along the stream channel. Other important functions of large wood in mid-order streams include the retention of salmon carcasses and organic detritus, which provide nutrients to the flora and fauna within the stream and in the adjacent riparian area (Bilby et al. 1996).

The role of large woody debris in high order streams is generally less well documented; however, historical records indicate that large debris jams once played a major role in floodplain and channel development on major rivers, such as the Willamette River in Oregon and other systems in the Puget Lowlands of Washington (Sedell and Luchessa 1981). In these high-order streams, large woody debris increased channel complexity by creating side channels, backwaters, and ponds, as well as refugia for aquatic organisms during winter storm events. During high flows, sediments were deposited on the floodplains and in riparian zones, increasing the productivity of these soils. Extensive snag removal and channelization over the last 100 years have diminished these roles of wood in larger river systems. Today, solitary pieces of woody debris are generally not large enough to span the active channel or substantially modify flows (Maser et al. 1988), although woody debris along the outside bends of river banks provides habitat in an otherwise simplified habitat zone (Swanston 1991). Wood snags that remain in the main channel are utilized by insects and fish, particularly in larger river systems with unstable sand substrate (Marzolf 1978; Benke et al. 1984).

Defining the zone of influence for input of all sources of large woody debris is difficult because methods of delivery differ. Most wood likely enters the stream from toppling or windthrown trees; however, wood may also enter the channel through mass wasting and debris torrents. The likelihood that a falling tree will enter the stream channel depends on tree height, distance from the stream channel, and the nature of the terrain. On level terrain, the direction that a tree will fall is essentially random (Van Sickle and Gregory 1990; Robison and Beschta 1990b) except along streambanks, where undercutting causes trees to lean and fall in the direction of the channel. On steep terrain, however, there is generally a higher probability that the tree will fall downslope into the stream channel. The greatest contribution of large wood to streams comes from trees within one tree height of the channel that topple into the stream (Figure 3-2) (FEMAT 1993). McDade et al. (1990) found that source distances of LWD were as far as 55 m from the stream channel in old-growth forests of the Coast and Cascade ranges (OR and WA) with average tree heights of 57.6 m. Murphy and Koski (1989) found that most (99%) large wood (pieces > 3 m in length) in southeastern Alaskan streams originated within 30 m of the channel (approximately 0.75 tree heights). For episodic inputs of large woody debris via mass wasting and debris torrents, defining the zone of influence becomes more difficult. The likelihood of wood entering the stream will vary with conditions that control the frequency of mass wasting, including the slope, soil type, and hydrology. Assessing appropriate zones of influence for these events is probably beyond our current level of scientific understanding. Cummins et al. (1994) and Reeves et al. (1995) report that O-order channels generate most landslides containing trees and coarse sediments.

3.9.6 Nutrients

Riparian zones mediate the flow of nutrients to the stream and are, therefore, important regulators of stream production. Subsurface flow from upland areas carries nutrients and dissolved organic matter to the riparian zone, where these materials are taken up by vegetation for plant growth or are chemically altered (Naiman et al. 1992). Lowrance et al. (1984) found that even narrow riparian zones along streams in agricultural lands significantly affected stream chemistry. Riparian forests modify the chemical composition and availability of carbon and phosphorus, and they promote soil denitrification through changes in the position of oxic-anoxic zones (Pinay et al. 1990 in Naiman et al. 1992). During overbank flows, nutrients from floodwaters may be absorbed by riparian vegetation, reducing the total nutrient load in the stream (Cummins et al. 1994). Dissolved organic matter inputs can occur from numerous sources besides groundwater. These

include leachate from entrained litter and large woody debris in the channel, algal, invertebrate, and fish excretions; and floodplain capture at the time of inundation (Gregory et al. 1991).

We found no published attempts to define zones of influence for nutrient cycling. Most likely, this reflects the difficulty in tracing the movement of nutrients, particularly with those elements such as nitrogen for which the number of alternative pathways is great. As discussed in Section 3.8, conditions throughout the watershed influence stream chemistry; consequently, the zone of influence extends to the top of the watershed, even though it may be years before nutrients ultimately find their way to the stream. However, the zone of most intense interaction is within the floodplain and hyporheic zones, where subtle changes in oxygen levels can dramatically affect nutrient composition and bioavailability.

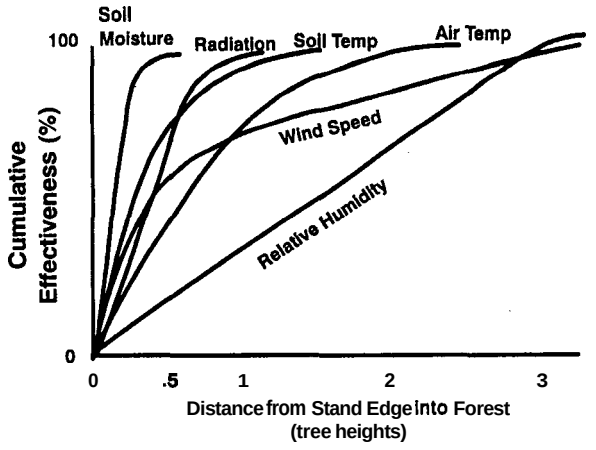


Figure 3-3. Riparian buffer effects on microclimate. From FEMAT (1993).

3.9.7 Microclimate

Although not well documented (O'Laughlin and Belt 1994), streamside vegetation can have a significant influence on local microclimates near the stream channel (FEMAT 1993). Chen (1991) reported that soil and air temperatures, relative wind speed, humidity, soil moisture, and solar radiation all changed with increasing distance from clear-cut edges in upslope forests of the western Cascades. Based on Chen's results, FEMAT (1993) concluded that loss of upland forests likely influences conditions within the riparian zone. FEMAT also suggested that riparian buffers necessary for maintaining riparian microclimates need to be wider than those for protecting other riparian functions (Figure 3-3).

3.9.8 Wildlife Habitat

Although riparian areas generally constitute only a small percentage of the total land area, they are extremely important habitats for wildlife. The attractiveness of riparian zones to wildlife likely reflects three attributes: the presence of water, which is essential to all life and generally scarce in the West (particularly east of the Cascade crest); local microclimatic conditions; and the more diverse plant assemblages found in riparian areas compared to surrounding uplands. The last characteristic derives from the dynamic nature of riparian zones, which typically leads to a mosaic of plant assemblages in different stages of ecological succession (Kauffman 1988). Brown (1985) reports that 87% of wildlife species in western Oregon and Washington use wetlands or riparian areas during some or all of their life cycle (FEMAT 1993). Thomas et al. (1979) found that 82% of all terrestrial vertebrates in the Great Basin of southeastern Oregon are either directly dependent on riparian zones or use riparian habitats more than any other habitat. Dependence of a majority of species on riparian zones has been demonstrated for all major vertebrate classes. Bury (1988) reported that 8 of 11 species of amphibians and 5 of 6 species of reptiles in Oregon either reside or breed in aquatic or riparian habitats. In northern California, approximately 50% of both reptiles and amphibians prefer riparian or aquatic habitats (Raedeke et al. 1988). Raedeke (1988) reviewed the published literature and found that 67% of native large mammals in the Pacific Northwest either depend on riparian areas or are more abundant in riparian areas than in surrounding uplands. Similar preferences for riparian habitat by small mammals, and especially bats, have also been documented (Cross 1988). Beschta et al. (1995) report that 55 species of birds in Oregon (approximately 46% of the total for which data were available) depend on or exhibit preferences for riparian habitats. For eastside ecosystems, the dependence of birds and other species on riparian zones is likely higher than for westside systems, where water and forests are more abundant.

3.10 Implications for Salmonids

The above discussion highlights the highly complex array of physical and chemical processes that occur across the landscape, in the riparian zone adjacent to streams and rivers, and within the stream channel. Large-scale geomorphic and climatic processes have together shaped the landscape of the Pacific Northwest, exerting dominant control over channel gradient and configuration. Although these

processes operate at scales of thousands to millions of years (Table 3-6), they are, nevertheless, important to consider in the development of salmonid conservation strategies. The current distribution of salmonids and other fish species in the Pacific Northwest is a direct consequence of tectonic activity dating back tens of millions of years that has both isolated and reconnected drainage basins through vertical lift and shifted subplates (McPhail and Lindsey 1986; Minckley et al. 1986). Similarly, climatic shifts and glaciation have alternately eliminated and stimulated reinvasion of fishes over significant portions of the Pacific Northwest landscape, as well as redistributed species into lower elevations or more southerly areas. Furthermore, the isolation of individual populations by geomorphic and glacial processes over time has allowed the evolution of unique stocks and species. Evolutionarily significant units (Waples 199lb) reflect the historical legacy wrought by geologic and climatic conditions over the millennia as well as adaptation to local environmental conditions that have prevailed since the last glacial period. Finally, long-term geomorphic and climatic processes together with hydrologic processes and vegetative cover, determine the rate at which nutrients, sediments, organic material, and water are transported from upslope areas into the stream channel. Consequently, the geomorphic and climatic setting determines the normal background rates of these processes, regulates the frequency and magnitude of natural episodic disturbances that reset and replenish streams, and govern the responses of specific watersheds to human perturbations.

Nested within this geomorphic and climatic context are a number of physical and chemical processes that further modify the landscape and that directly influence stream channel characteristics and water chemistry. These processes include surface erosion, landslides, floods, debris torrents, ice flows, droughts, beaver activity, and wildfire, and they operate at ecological times scales—generally from days to decades or centuries—regulating the input of sediment, nutrients, and organic material to the stream (Table ***3-6)***. The riparian zone acts as the interface between terrestrial and aquatic ecosystems, moderating the effects of upslope processes as well as providing other critical functions (e.g., shading, bank stabilization, nutrient transformation, allochthonous inputs). Together, these processes determine macrohabitat characteristics, such as general channel morphology and pool-riffle sequences as well as microhabitat characteristics, such as depth, velocity, cover, temperature, and substrate.

The processes that influence salmonid habitats may be either cyclical in their occurrence (e.g., seasonal temperature, streamflow, and leaf-fall patterns), or episodic in nature (e.g., wildfires, landslides, floods, debris torrents). It is critical to recognize that these cycles or disturbances are fundamental and vital parts of ecosystem function, even though they may be temporarily disruptive of aquatic ecosystems. Studies of geomorphology and paleoecology indicate that disturbance is continual, sometimes across large areas, and often unpredictable. In eastside ecosystems the changes are most often associated with climatic changes that render vegetation more susceptible to disturbances such as fire and disease (Johnson et al. 1994). Eastside forests and rangelands have evolved with periodic disturbances, and when they do not receive them, they become increasingly unstable (Henjum et al. 1994; Johnson et al. 1994). If drought or fire do not alter these systems, then disease or insects will. Naturally occurring mass-soil movements and erosion introduce large woody debris, rock, gravel, and fine materials into stream channels, substantially modifying conditions for salmonids. Floods and debris torrents are dominant disturbances affecting westside stream systems (Swanston 1991) and may significantly alter channel morphology, scouring channels and creating debris jams and coarse sediment deposits that eventually produce important spawning and rearing areas for salmonids.

Salmonids have evolved not only to the general conditions that are typical of a watershed, but to the specific disturbance regimes found in that watershed. Human activities potentially modify disturbance regimes in three distinct ways: by increasing the frequency of disturbance events, by altering the magnitude of these events, and by affecting the response of the stream channel to disturbance events through modification of instream characteristics. Sediment delivery, for example, is essential to the development and maintenance of spawning gravels for salmonids. However, alteration in sediment composition, delivery rates, or fate can be damaging to salmonids, resulting in the degradation of spawning gravels and rearing habitats. Similarly, floods and droughts are important determinants of fish assemblage structure; however, increases in the frequency of these events may result in population declines, shifts in community structure, and decreases in biodiversity. The effects of human-caused alterations on salmonids and their habitats are discussed in greater detail in Chapter **6** of this document.

Table 3-6. Approximate ranges of recurrence of landscape and channel-forming processes and the effects of these events on stream habitats. Modified from Swanston (1991). Reproduced with permission from the publisher.

Event	Range of recurrence (years)	Channel changes	Habitat effects
Tectonics	1,000 – 1,000,000	Creation of new drainages; major channel changes including stream capture because of regional upwarping and faulting.	Subsidence in alluvial and coastal fills creating zones of deposition with increased fines; Steep erosive channels caused by upwarping leads to coarser sediments.
Climatic change	1,000 – 100,000	Major changes in channel direction; major changes in channel grade and configuration; valley broadening or downcutting; alteration of flow regime.	Changes in type and distribution of spawning gravels; changes in frequency and timing of disturbance events; shifts in species composition and diversity.
Volcanism	1,000 – 100,000	Local blocking and diversion of channel by mudflows and tephra; valley filling and widening; major changes in channel grade and configuration.	Changes in type and distribution of spawning gravels. Major inputs of sand and silt from tephra.
Slumps and earthflows	100 – 1,000	Low-level, long term contributions of sediment and large woody debris to stream channels; partial blockage of channel; local base level constriction below point of entry; shifts in channel configuration.	Siltation of spawning gravels; scour of channel below point of entry; accumulation of gravels behind obstructions; partial blockage of fish passage; local flooding and disturbance of side-channel rearing areas.
Wildfire	1 – 500	Increased sediment delivery to channels; increased large woody debris in channels; loss of riparian vegetation cover; decreased litterfall; increased channel flows; increased nutrient levels in streams.	Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and over-wintering habitat; decreased availability of fine woody debris; increased availability of food organisms.
Windthrow	10 – 100	Increased sediment delivery to channels; decreased litterfall; increased large woody debris in channel; loss of riparian cover.	Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and over-wintering habitat; decreased fine organic debris.
Insects and disease	10 – 100	Increased sediment delivery to channels: loss of riparian vegetation cover; increased large woody debris in channels; decreased litterfall.	Increased sedimentation of spawning and rearing habitat; increased summer temperatures; decreased winter temperatures; increased rearing and overwintering habitat.

Table 3-6. Approximate ranges of recurrence of landscape and channel-forming processes and the effects of these events on stream habitats. Modified from Swanston (1991). Reproduced with permission from the publisher.

Event	Range of recurrence (years)	Channel changes	Habitat effects
Activities of beavers	5 – 100	Channel damming; obstruction and redirection of channel flow; flooding of banks and side channels: ponding of streamflow; siltation of gravels behind dams.	Improved rearing and overwintering habitat; increased water volumes during low flows; slack-water and back-water refuge areas during floods; refuge from reduced habitat quality in adjoining areas; limitation on fish migration: elevated water temperatures; local reductions in dissolved oxygen.
Debris avalanches and debris torrents	5 – 100	Large, short-term increases in sediment and large woody debris contributions to channel; channel scour; large-scale movement and redistribution of bed-load gravels and large woody debris; damming and obstruction of channels; accelerated channel bank erosion and undercutting; alteration of channel shape by flow obstruction; flooding.	Changes in pool to riffle ratio; shifting of spawning gravels: siltation of spawning gravels; disturbance of side-channel rearing areas; blockage of fish access; filling and scouring of pools and riffles; formation of new rearing and overwintering habitat.
Major storms; floods; rain-on-snow events	1.0 – 10	Increased movement of sediment and woody debris to channels; flood flows; local channel scour; movement and redistribution of coarse sediments; flushing of fine sediments: movement and redistribution of large woody debris.	Changes in pool to riffle ratio; shifting of spawning gravels; increased large woody debris jams; siltation of spawning gravels; disturbance of side-channel rearing areas; increased rearing and overwintering habitat; local blockage of fish access: filling and scouring of pools and riffles.
Seasonal precipitation and discharge: moderate storms: freezing and ice formation	0.1 – 1.0	Increased flow to bank-full width; moderate channel erosion; high base-flow erosion: increased mobility of in-channel sediment and debris; local damming and flooding; sediment transport by anchor ice; gouging of channel bed; reduced winter flows.	Changes in pool to riffle ratio; siltation of spawning gravels; increased channel area; increased access to spawning sites; flooding of side-channel areas; amelioration of temperatures at high flows; decreased temperatures during freezing; dewatering of gravels during freezing; gravel disturbance by gouging and anchor ice.
Daily to weekly precipitation and discharge	0.01 – 0.1	Channel width and depth; movement **and** deposition of fine woody debris; fine sediment transport and deposition.	Minor siltation of spawning gravels; minor variation in spawning and rearing habitat; increased temperature during summer low flows.

4 Biological Processes and Concepts

The physical and chemical environment of aquatic ecosystems forms the template upon which biological systems at all levels are organized (Southwood 1977; Poff and Ward 1990). The spatial and temporal patterns of water quantity and velocity, temperature, substrate, and dissolved materials influence the physiology and behavior of organisms, the dynamics and evolution of populations and metapopulations, and the trophic structure and diversity of aquatic communities. Modification of physical and chemical attributes of water bodies through land-use and water-use practices, and direct alteration of specific biological components of these systems, can result in changes to individual organisms, populations, and communities. In this section, we briefly review some fundamental biological processes that occur in aquatic ecosystems, focusing on those processes that are likely to be affected by modifications to physical and chemical habitat characteristics.

4.1 Organism Level

The survival of salmonids in the wild depend on their ability to carry out basic biological and physiological functions including feeding, growth, reproduction, respiration, hydromineral balance, smoltification (anadromous forms), and migration. The fate of populations and the outcome of higher-level biological interactions—competition, predation, and disease—ultimately depend on the performance of individuals in obtaining food, defending space, maintaining physiological health, or otherwise coping with their ecological circumstances. Characteristics of the physical and chemical environment of fish, particularly water temperature, regulate the rates at which these processes occur. A detailed discussion of the complex interactions between fish and their environments is beyond the scope of this document; however, a brief review of the fundamental biological processes is essential to understanding how habitat modifications may affect salmonids.

4.1.1 Feeding and Growth

Juvenile salmonids are generally opportunistic in their feeding habits while in freshwater, primarily consuming drifting aquatic or terrestrial invertebrates in streams, and macroinvertebrates and zooplankton in lakes and estuaries. Bull trout especially, as well as resident rainbow and cutthroat trout, may feed on other fishes and amphibians during their adult stages, particularly in systems where they attain large sizes. A summary of specific dietary items for anadromous and resident salmonids can be found in Meehan and Bjornn (1991); a more detailed examination of dietary habits of Pacific salmon can be found in Groot and Margolis (1991).

Environmental conditions influence the demand for food, the amount and type of prey available to salmonids, the ability of fish to capture prey, and the costs of obtaining food. Ingestion rates of fishes generally increase with increasing temperature, except when temperatures exceed the thermal optimum for the species (Brett 1971). Low levels of dissolved oxygen may also lead to suppression of appetite in salmonids (Jobling 1993). Increased levels of sediment may alter substrate composition, filling substrate interstices, and thereby affecting the total abundance and composition of invertebrate prey. Similarly, reductions in fine and coarse organic litter inputs can both reduce the food base and alter habitat structure for prey organisms. Turbidity in streams may reduce light penetration, decreasing the reactive distance of salmonids to prey and limiting production of benthic algae. Nutrient availability also affects total food availability by controlling primary production. For stream-dwelling salmonids, the energetic costs associated with acquiring food depend on current velocities at holding and feeding stations. Many salmonids seek out areas of slow water velocity immediately adjacent to faster waters, presumably because these areas provide greater food per unit of energy expended in maintaining position (Smith and Li 1983; Fausch 1984). Heterogeneity of velocity, therefore, creates microhabitats that are energetically favorable. All of these factors can be affected by human alterations of habitat or watershed processes, fundamentally affecting the ability of individuals to satisfy their food intake requirements.

Once food energy is consumed, it is used in a variety of metabolic processes. These include respiratory and circulatory processes that deliver oxygen to various tissues, maintenance of cells, digestion of food, assimilation and storage of nutrients, and various muscular activities (e.g., swimming and other behaviors). After satisfying

these demands, surplus energy that is not excreted as waste can be devoted to growth of body and reproductive tissues. A number of environmental variables influence the growth rate of salmonids. In general, growth rates increase with increasing temperature up to a thermal optimum, above which reductions in appetite and increasing metabolic demands combine to reduce the growth rate. Growth rates of salmonids, as well as food conversion efficiency, may also be reduced when dissolved oxygen levels below 5–8 mg/L or 60%–70% of saturation (Jobling 1994). Other chemical factors that influence growth rate include ammonia and salinity (Moyle and Cech 1982), as well as various other pollutants. Because water velocity determines metabolic demands of fish, it indirectly determines how much energy is available for anabolic processes. Thus, human-caused changes in water quality, natural flow regimes, or hydraulic characteristics all may inhibit growth and development of salmonids.

4.1.2 Reproduction and Embryological Development

Energy reserves of salmonids must be sufficient to allow for gamete production after growth and metabolic costs are incurred. Anadromous salmonids have particularly high energy requirements because they must have sufficient reserves to undergo lengthy migrations and negotiate barriers in order to reach the ocean and then return to their spawning tributaries. Modifications of temperature, water quality, streamflow, and physical structure all affect how much energy can be devoted to reproductive output. The development of embryos and alevins in the gravel is affected by several environmental factors. Water temperature greatly influences times to hatching and emergence for Pacific salmonids. Development time decreases in an asymptotic fashion with increasing incubation temperatures with the rate of change in development time relative to temperature increase being greatest at the low end of the tolerable temperature range (Beacham and Murray 1990). Consequently, small increases in temperature at the low end of the range can substantially alter the time of hatching and emergence of salmonids. Early emergence because of warming of water temperatures may increase exposure of fry to high-flow events and alter the natural synchrony between emergence and predator cycles or prey cycles. Scrivener (1988) found that chum salmon in Carnation Creek emerged and migrated to sea 4 to 6 weeks earlier after logging compared with prelogging years in response to water temperature increases. In a companion study, Holtby (1988) reported that coho salmon emerged up to 6 weeks early in response to logging. Temperatures may influence the size of emerging fry. For example, coho salmon reared at 4°C were larger than those reared at warmer temperatures (Beacham and Murray 1990). In contrast, fry of pink salmon tended to be larger when reared at 8°C than when reared at 4°C.

Dissolved oxygen concentrations in redds also influence the survival and development rate of embryos and alevins, as well as the size of emerging fry (Warren 1971). Streamflow may regulate the flow of water through redds and hence the levels of dissolved oxygen. Environmental changes, such as siltation or altered flow regimes, that reduce the flow of water can thus adversely affect embryo and alevin development and survival. Silt concentration in gravels may also impede emergence of fry. Phillips et al. (1966) found that emergence of coho fry decreased as the percentage of fine sediments in the gravel increased, presumably because of reduced oxygen content and increased difficulty of fry in reaching the surface. Increased frequency of high scouring flows or debris torrents, which are associated with disturbed catchments (Swanston 1991), may further affect egg and alevin survival.

4.1.3 Respiration

Most of the energy used by salmonids to swim, locate food, grow, and reproduce is provided through metabolic processes that require oxygen. Because water contains only about 3.3% of the amount of oxygen contained in air, the efficient extraction of oxygen is critical to survival. Fish, and salmonids in particular, have evolved elaborate gill structures that facilitate the uptake of oxygen for delivery to other parts of the body. Environmental conditions can have a significant influence on the oxygen demands of fish, the amount of oxygen present in water, and the ability of fish to take up that oxygen. In general, the oxygen demands increase with increasing temperature, although oxygen consumption may decrease as temperatures approach lethal levels, particularly at high levels of activity (Brett 1971). In contrast, dissolved oxygen levels in saturated water are inversely proportional to temperature with water at 5°C holding approximately 30% more oxygen than water at 20°C. Oxygen demand is also influenced by water velocity, which determines the swimming speed required of salmonids to maintain their position in the current. High levels of suspended solids in water may influence respiration by abrading or clogging gill surfaces (Warren 1971). Similarly, pollutants can cause mucous secretions to coat gill surfaces, inhibiting the exchange of oxygen. Excessive amounts of algae and easily decomposable organic material in water increases plant and microbial oxygen demand, thereby decreasing dissolved oxygen concentration. Low levels of dissolved oxygen, in turn, impede the ability of hemoglobin within the blood to bind with oxygen,

effectively reducing the delivery of oxygen to body tissues (Moyle and Cech 1982). On the other hand, gas supersaturation from dam spills and intense algal photosynthesis can create gas bubbles in fish gills and tissue, resulting in decreased respiratory efficiency, disease, or death. All of these factors can influence the ability of fish to satisfy their oxygen demands.

4.1.4 Smolfification

The transition from fresh to salt water marks a critical phase in the life history of anadromous salmonids. Emigration to the ocean is preceded by rapid physiological, morphological, and behavioral transformations that preadapt fish for life in salt water and initiate their downstream movement (Folmar and Dickhoff 1980; Wedemeyer et al. 1980; Groot 1982). Once at sea, newly arrived smolts must acclimate to a markedly different set of ecological circumstances, including new food resources, new predators, and a substantially different physical environment. Much of the total ocean mortality incurred by salmon smolts is believed to occur during this period of early ocean life (Manzer and Shepard 1962; Matthews and Buckley 1976; Walters et al. 1978; Fisher and Pearcy 1988; Pearcy 1992). Consequently the timing of ocean entry is likely adaptive to maximize survival or growth (Miller and Brannon 1981; Riddell and Leggett 1981; Murphy et al. 1988; Beacham and Murray 1990).

Because development and growth are highly influenced by water temperatures, modifications to thermal regimes can potentially alter the time of smoltification (reviewed in Wedemeyer et al. 1980; Hoar 1988). Similarly, temperature and streamflow patterns may be important cues for releasing migratory behavior (Hoar 1988). Consequently, alterations in normal hydrologic and thermal patterns may trigger movement into the ocean at times that are less favorable for growth and survival. The parr-smolt transformation may also be affected by exposure to contaminants, including heavy metals, which alter enzymatic systems involved in osmoregulation and may inhibit migratory behavior (Wedemeyer et al. 1980). Structural alterations that hinder salmonids during the smolt transformation include loss of large woody debris and habitat complexity in streams and estuaries, which reduces cover and food supplies during this critical period.

4.1.5 Summary

A useful way of summarizing the effects of environmental factors on individual fish is through a simple energy budget. Food energy that is ingested by fish (I) has several potential fates. It is either expended during metabolic processes (M), deposited as new somatic (body) or reproductive tissue (G), or excreted as waste products (E) (Jobling 1993). Thus the energy balance can be expressed as

$$I = M + G + E \ .$$

Environmental conditions influence all aspects of a fish's energy budget. Temperature, in particular, has pervasive effects on bioenergetic pathways, affecting appetite, digestion rate, standard and active metabolic rates, and food conversion efficiency (i.e., the proportion of food energy absorbed by the fish). Because the energetic costs of swimming increase exponentially with increasing speed (Jobling 1993), water velocity determines how much energy is expended in maintaining position and obtaining food. Dissolved oxygen concentrations also affect food consumption and metabolic processes (Warren 1971), as do various chemical pollutants. These and other environmental factors interact to determine the amount of energy expended on metabolic processes, and hence determine the energy left over for growth and reproduction. When changes in environmental conditions reduce the amount of food available or alter the efficiency with which food is captured and assimilated, the performance of individual fish declines. This reduction in performance, in turn, affects the outcome of higher-level interactions including competitive, predator-prey, and disease-host relationships (see Section 4.3).

4.2 Population Level

Salmonid populations are noted for their complex life cycles, diverse life histories, and tendency to form locally adapted stocks. The interaction among various subpopulations (metapopulation dynamics) has important implications for conservation.

4.2.1 Generalized Life Cycle

The life cycle of anadromous salmonids consists of several distinct phases, at least three of which involve significant shifts in habitat. Adult salmon migrate from the ocean into their natal stream to spawn. Females construct a "redd" in the stream gravel into which eggs are deposited, fertilized by males, and subsequently covered with gravel. All adult salmon die after spawning, usually within a few weeks. Females will typically spend one to three weeks guarding the redd site before dying, whereas males may seek out and spawn other females. The fertilized embryos develop for a period of one to several months, depending on temperature and dissolved oxygen availability, before hatching occurs. The emergent "alevins" remain in the gravel, nourished by a yolk sac, for another few weeks to a month or more. Once yolk-sac absorption is complete, the fry emerge from the gravel and begin actively feeding on drifting material. The period of freshwater rearing lasts from a few days to several

years, depending on the species, after which juveniles undergo smoltification. Smolts migrate to the ocean, where the majority of growth occurs, before returning to spawn as adults, completing the cycle (Figure 4-1).

The life cycles of the anadromous trout and char differ from those of salmon in that some adults may survive after spawning, migrate back to the ocean, and return to spawn a second or third time. Resident salmonids, including kokanee salmon (i.e. landlocked sockeye salmon), bull trout, cutthroat trout, rainbow trout, and mountain whitefish, do not have an oceanic phase but commonly undergo substantial migrations to and from rearing areas in lakes or larger rivers. With the exception of kokanee salmon, which die after spawning, the resident forms usually spawn multiple times over their lifetimes.

4.2.2 Life History

Although all anadromous salmonids share the general life cycle discussed above, substantial differences exist in the period of time that the different species spend in freshwater and marine environments (Table 4-1), and the types of habitat they use for spawning and rearing. In addition, a high degree of variation in life histories can exist within each species.

Life-History Patterns

Extensive reviews of the life histories and general habitat preferences of trout, char, and Pacific salmon can be found in Groot and Margolis (1991) and Meehan and Bjornn (1991) from which much of the information below was taken. Pink and chum salmon typically spawn in coastal streams not far from tidewater—chum occasionally within the tidal zone—and have the shortest freshwater phase, entering the ocean soon after they emerge from the gravel. Almost without exception, pink salmon mature at 2 years of age, at which time they return to freshwater to spawn. Chum salmon are more variable, spending from 2 to 5 years in the ocean before returning to their natal area to spawn. Coho salmon generally spawn in small, low-gradient streams or stream reaches in both coastal and interior systems. Juveniles typically spend from 1 to 3 years in freshwater; however, in the southern portion of their range (including Washington, Oregon, and California) most fish migrate to sea after just one year. Adults return to spawn after approximately 18 months at sea, although "jack" males may return after only six months in the ocean (Sandercock 1991). The life histories of sockeye and chinook salmon are more variable. Sockeye salmon most often spawn in the inlet or outlet streams of lakes. Shortly after emergence, sockeye fry migrate into these lakes, where they reside for 1 to 3 years. Juveniles then migrate to the ocean, where they spend 2 to 3 years. Chinook salmon generally spawn in small to medium-sized rivers, but may also spawn in large river systems such as the mainstem Columbia. Chinook salmon display two dominant life-history types, an ocean type that is typical of fall-run stocks and a stream type that is characteristic of spring-run fish. Those exhibiting the ocean-type life history usually spend only a few months in freshwater before migrating to sea. Stream-type fish spend 1 to 2 years in freshwater. Both ocean- and stream-type fish can reside anywhere from 2 to 5 years in the ocean, although jacks may spend less than a year at sea before returning to spawn. Within any given population, multiple life-history patterns may be observed. Based on time of freshwater and estuarine residence, Reimers (1973) identified five distinct life-history patterns for fall chinook salmon in the Sixes River, Oregon.

The anadromous trout and char, including steelhead and cutthroat trout, and Dolly Varden, exhibit considerable life-history variation as well (Table 4-1). Steelhead trout tend to spawn in small streams and favor relatively high-gradient reaches. Freshwater residence can last from 1 to 4 years, while ocean residence ranges from a few months ("half-pound") males to 4 years. Although most adult steelhead die after spawning, up to 30% may live to return to the ocean and spawn again in subsequent years, particularly in coastal streams where the spawning migrations are fairly short (Meehan and Bjornn 1991). Consequently, the number of potential life-history types is large. Anadromous cutthroat trout most commonly spawn in small headwater streams and spend 2 to 4 years in freshwater before migrating to the ocean during the spring, where they generally remain until the next fall. As with steelhead trout, some adults may live after spawning, migrate back to the ocean, and return a second or third time. Dolly Varden spawn in coastal streams and exhibit complex life-history patterns. Juveniles typically rear in higher-velocity habitats for several years (Meehan and Bjornn 1991). After smoltification, Dolly Varden enter the ocean, but may repeatedly return to freshwater habitats during the winter months to rear in lakes, sometimes away from their natal areas. Thus, it is difficult to generalize about the periods of freshwater and marine residence for Dolly Varden.

Resident trout, char, and whitefish spend their entire lives in freshwater; however, life-history patterns may still be quite diverse (Table 4-1). Varley and Gresswell (1988) identified four principal life-history patterns for Yellowstone Lake cutthroat trout: fluvial populations that remain in their natal streams throughout their lives, fluvial-adfluvial populations that reside in larger rivers but spawn in small tributaries, lacustrine-adfluvial populations that

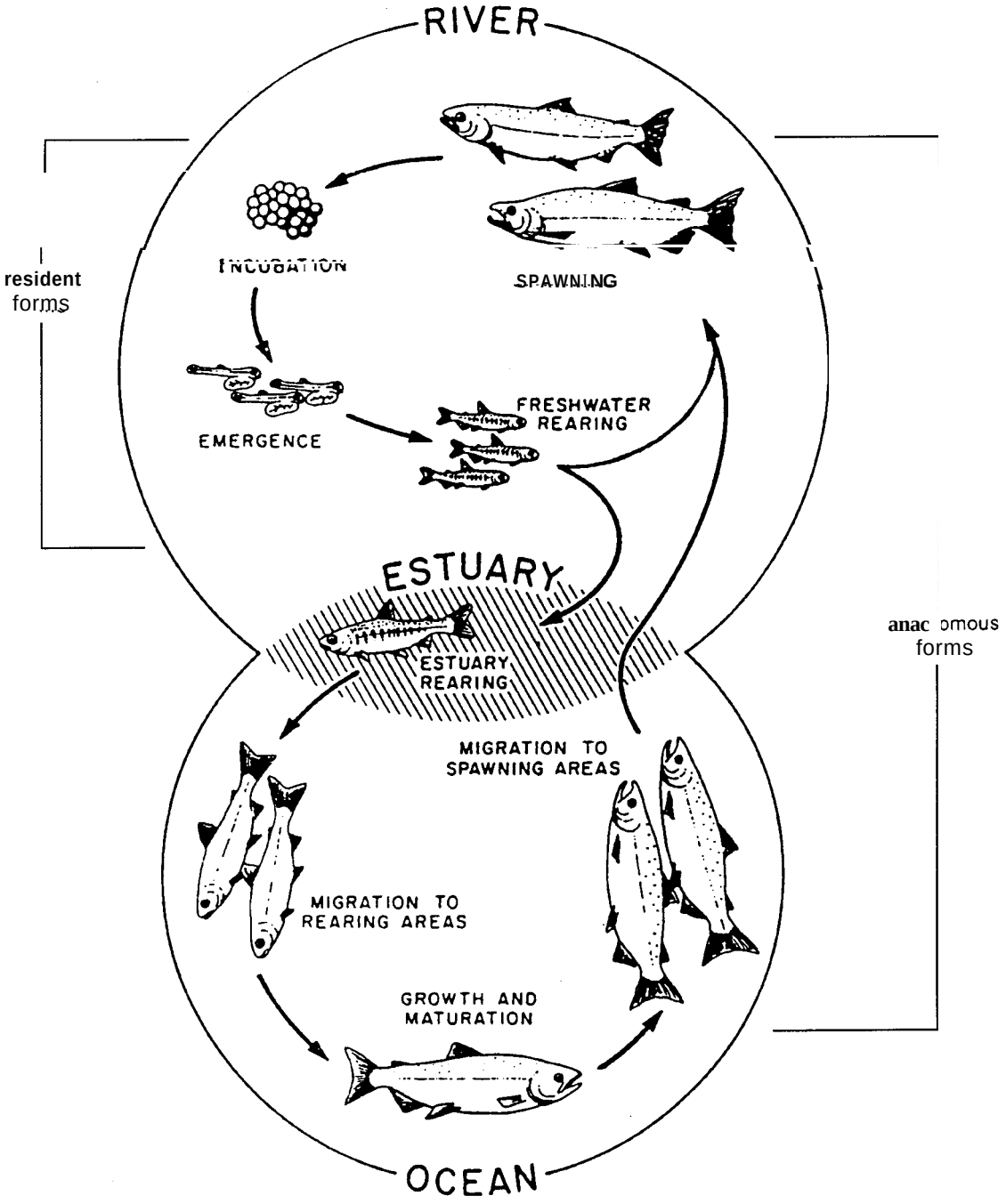


Figure **4-1.** Generalized salmonid life cycle, showing freshwater and ocean components. Modified from Nicholas and Hankin (1988). Reproduced with permission from the authors.

Table 4-1. Life histories of Pacific salmonids.*

Species	Spawning migration	Spawning period	Spawning area	Life history	Most common age at maturity†
ANADROMOUS SALMON					
Chum salmon	Summer to winter	Summer to winter	Usually near tidewater	Fry go directly to sea. 2 – 5 years ocean.	4_1
Pink salmon	Late summer to early fall	Late summer to early fall	Usually near tidewater	Fry go directly to sea. 2 years ocean.	2_1
Sockeye salmon	Spring to fall	Late summer to fall	Tributaries of lakes	1 – 3 years lake. 2 – 3 years ocean.	4_2 5_3
Coho salmon	Summer to fall	Fall to early winter	Small headwater streams	1 – 3 years FW.‡ 6 months jack. } 18 months adult } ocean.	3_2
Chinook salmon	Spring to fall	Summer to early winter	Large rivers	3 months – 2 years FW. 2 – 5 years ocean.	4_1 (Ocean) 5_1 (Stream)
ANADROMOUS TROUT AND CHAR					
Steelhead trout	Summer to winter	Late winter to spring	Small headwater streams	2 – 3 years FW. 1 – 3 years ocean. Repeat spawners.	mature 4 – 5
Searun cutthroat trout	Fall to winter	Late winter to early spring	Small headwater streams	2 – 4 years FW. 2 – 5 months ocean. Repeat spawners.	mature 3 – 4
Dolly Varden	Late summer to fall	Fall	Main channels on rivers	2 – 4 years FW. 2 – 4 years ocean. Repeat spawners.	mature 5 – 6 die 6 – 7
RESIDENT SPECIES					
Kokanee salmon	Late summer to fall	Late summer to fall	Tributaries of lakes, lakeshores	Juveniles migrate to lakes to reside.	3 – 4
Rainbow trout	Spring	Spring	Small headwater streams	Fluvial, adfluvial, lacustrine-adfluvial life histories. Variable residence in natal streams, rivers, & lakes.	2 – 3
Cutthroat trout	Spring	Spring to early summer	Small headwater streams	Fluvial, adfluvial, lacustrine-adfluvial life histories. Variable residence in natal streams, rivers, & lakes.	3 – 4
Bull trout	Fall	Fall	Large streams with ground water infiltration	Juveniles migrate from tributaries to lakes or larger streams at about 2 years, highly variable.	4 – 9
Mountain whitefish	Fall	Fall	Mid-sized streams, lakes	Reside in streams and lakes.	3 – 4

* Data from Groot and Margolis (1991); Meehan and Bjornn (1991); Pratt (1992); Behnke (1992); and Moyle (1976).
† Gilbert-Rich age designation in years.
‡ FW = freshwater.

reside in lakes and spawn in tributary streams, and allucustrine populations that reside in lakes and migrate down outlet streams to spawn. Rainbow trout may spawn in streams, in lake inlets or outlets, or in lake springs, and rear in streams or lakes (Behnke 1992). Bull trout reside in a variety of freshwater habitats including small streams, large rivers, and lakes or reservoirs (Meehan and Bjornn 1991). Some populations spend their entire lives in cold headwater streams. In other populations, juveniles spend from 2 to 4 years in their natal stream before migrating into lakes or reservoirs, where they reside for another 2 to 4 years before returning to their natal stream to spawn. Mountain whitefish spawn in streams and rivers and reside there throughout their lives although substantial migrations from larger rivers into smaller spawning tributaries are common.

Implications of Life-History Diversity for Salmonid Conservation

The remarkable diversity of life histories exhibited by Pacific Northwest salmonids reflects adaptation to a wide array of habitats. As a group, the salmonids inhabit streams ranging from mountain headwaters to large lowland rivers, in regions varying from arid and semiarid shrublands to temperate rainforests. Reproduction may occur in streams, lakes, or intertidal sloughs; rearing of juveniles occurs in streams and lakes for some species and in estuaries and oceans for others (Table 4-2). In any particular habitat, spatial and temporal differences in micro- and macrohabitat utilization permit the coexistence of ecologically similar species (Everest et al. 1985). Within species, life-history diversity allows salmonids to fully utilize available freshwater, estuarine, and ocean environments. Species that occupy several habitat types, or that have multiple freshwater and marine residence times, effectively spread ecological risk (sensu Den Boer 1968) such that the impacts of environmental fluctuation on populations are distributed through time and space. Consequently, species are likely to differ in their response to human-caused perturbation. The diversity among species and by life stage indicates that most accessible freshwater habitats are used year round if environmental conditions are suitable (Table 4-3).

Life-history diversity should be considered in the development of salmonid conservation strategies and local enhancement measures (Carl and Healey 1984; Lichatowich et al. 1995). The historically high abundance of salmonids in the Pacific Northwest was due in part to the diversity of life-history patterns exhibited by the various species. Habitat simplification through land-use and water-use practices has effectively simplified this diversity in life-history organization. In the Columbia River, for example, salmonids of various species and life stages were present in the mainstem year round. Because of alteration of temperatures and flow regimes, the temporal usage of the mainstem and major tributaries is now far more restricted. Historical records indicate that the Yakima River once supported **six** life-history types of spring chinook salmon, two of which reared in warmer, low-elevation, mainstem reaches. Today, because irrigation withdrawals have reduced flows and increased temperatures, the population consists only life-history types that rear in upper tributaries—the life-history types that utilized the lower mainstem for rearing have been eliminated (Lichatowich et al. 1995). Restoration of such populations to a harvestable level will require restoration of habitat conditions suitable for all life-history types of chinook salmon. Differences in life histories also affect the response of salmonids to harvest. Salmon that spend several years at sea before maturing are more vulnerable to troll fisheries than those that spend only a year at sea (see Section 6.11).

4.2.3 Stock Concept and Local Adaptation

Among the most remarkable characteristics of anadromous salmonid species is their tendency to return to their natal stream to spawn during a particular season often after ocean migrations of a thousand miles or more. Although the strong homing tendency of salmonids is most conspicuous in anadromous species, it may be common in resident populations as well. Lake-dwelling populations of cutthroat and bull trout that spawn in tributaries have also been shown to return to their natal stream to spawn with low rates of straying (Pratt 1992; Gresswell et al. 1994), and it is likely that stream-dwelling residents also display some fidelity to their natal area. As a consequence of homing, salmonid species typically comprise numerous local populations or "stocks" that are to varying degrees reproductively isolated from other such populations. Ricker (1972) defined a stock as "the fish spawning in a particular lake or stream (or portion of it) at a particular season [that] to a substantial degree do not interbreed with any group spawning in a different place, or in the same place at a different season."

The homing and resultant reproductive isolation of stocks provide a mechanism by which local populations become uniquely adapted to the specific suite of environmental conditions encountered during their life histories. Ricker, in his classic 1972 paper that formalized this concept, catalogued dozens of examples of local variation in morphological, behavioral, and life-history traits and provided evidence that many of these traits are to some degree heritable. For a trait to be considered adaptive, it must not only be differentially expressed, but it must

Table 4-2. Variation in life histories of Pacific salmonids. Modified from Everest et al. (1985).

Species (race)	Life histories	Spawns in			Rears in			
		Lakes	Streams	Intertidal	Lakes	Streams	Estuaries	Ocean
Pink salmon	Anadromous		X			X	X	X
	Anadromous		X					X
	Anadromous			X			X	X
Chum salmon	Anadromous		X			X	X	X
	Anadromous		X			X		X
	Anadromous		X					X
	Anadromous			X			X	X
Coho salmon	Anadromous		X			X	X	X
	Anadromous		X			X		X
Sockeye salmon	Anadromous		X		X			X
	Anadromous	X			X			X
Kokanee salmon	Resident		X		X			
	Resident	X			X			
Chinook salmon (spring)	Anadromous		X			X	X	X
	Anadromous		X			X		X
Chinook Salmon (fall)	Anadromous		X			X	X	X
	Anadromous		X			X		X
Cutthroat trout	Resident		X			X		
	Resident		X		X			
Cutthroat trout (searun)	Anadromous		X			X	X	X
	Anadromous		X			X		X
Rainbow trout	Resident		X			X		
	Resident		X		X			
	Resident	X			X			
Steelhead trout	Anadromous		X			X		X
Bull trout	Resident		X			X		
	Resident		X		X			
Dolly Varden	Anadromous		X			X	X	X
	Anadromous		X			X		X
	Anadromous		X		X			X
Mountain whitefish	Resident		X			X		
	Resident	X			X			

Table 4-3. Seasonal occurrence of adult, embryonic, and juvenile anadromous salmonids in freshwaters of western Oregon and Washington. From Everest et al. (1985).

Species	Life Stage	Months											
		Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec
Pink salmon	Adult						■	■	■	■	■		
	Young	■	■	■	■								■
	Eggs	■	■	■				■	■	■	■	■	■
Chum salmon	Adult	■	■	■						■	■	■	■
	Young	■	■	■	■	■							
	Eggs	■	■	■	■					■	■	■	■
Coho salmon	Adult	■	■								■	■	■
	Young	■	■	■	■	■	■	■	■	■	■	■	■
	Eggs	■	■	■								■	■
Sockeye salmon	Adult	■					■	■	■	■	■	■	■
	Young	■	■	■	■	■	■	■	■	■	■	■	■
	Eggs	■	■	■						■	■	■	■
Spring chinook salmon	Adult			■	■	■	■	■	■	■	■		
	Young	■	■	■	■	■	■	■	■	■	■	■	■
	Eggs	■	■								■	■	■
Fall chinook salmon	Adult	■							■	■	■	■	■
	Young	■	■	■	■	■	■	■	■	■			
	Eggs										■	■	■
Sea-run cutthroat trout	Adult	■	■	■						■	■	■	■
	Young	■	■	■	■	■	■	■	■	■	■	■	■
	Eggs		■	■	■								
Winter steelhead trout	Adult	■	■	■	■	■	■				■	■	■
	Young	■	■	■	■	■	■	■	■	■	■	■	■
	Eggs	■	■	■	■	■							■
Summer steelhead trout	Adult	■	■	■	■	■	■	■	■	■	■	■	■
	Young	■	■	■	■	■	■	■	■	■	■	■	■
	Eggs	■	■	■	■								■
Dolly Varden	Adult	■	■	■	■	■	■	■	■	■	■	■	■
	Young	■	■	■	■	■	■	■	■	■	■	■	■
	Eggs									■	■	■	■

confer some advantage to the individuals exhibiting that trait. More formally, Taylor (1991) defined local adaptation as "a process that increases the frequency of traits within a population that enhance the survival or reproductive success of individuals expressing such traits." He identified many examples of variation in morphological, behavioral, developmental, biochemical, physiological, and life-history traits in the family Salmonidae that are both heritable and believed to be adapted to local conditions. Results from his extensive review are summarized in Table **4-4.**

Despite the fact that the stock concept is generally credited to Ricker, the implications of stock formation and local adaptation in conservation of salmonid species have long been recognized. Rich (1939) proposed that conservation of a species that is made up of numerous, isolated, self-perpetuating units depends on conserving each constituent part. While Rich argued that local adaptation was not necessary for stocks to be the appropriate unit of management, the recognition that stocks do differ in heritable traits and that these differences are a consequence of differential selection serves to strengthen the argument for conserving individual salmonid stocks. The loss of local stocks changes the genetic composition and reduces the genetic variability of the species as a whole (Nehlsen et al. 1991), reducing its ability to respond to environmental change.

From Table 4-4 it is evident that many traits of salmonids are adaptations to environmental conditions that may be significantly altered by human activities. In the wake of rapid and extensive anthropogenic change, traits that were once adaptive may be rendered maladaptive. For example, the timing of spawning, emergence, and smoltification of salmonids are clearly linked to stream temperature regimes as are development rates of eggs and juveniles. Warming of stream temperatures through loss of riparian canopy, releasing water from reservoirs, or using irrigation practices can advance development or alter the timing of life-history events and potentially disrupt natural synchronies in biological cycles that have evolved over thousands of years. Alteration of temperatures may also affect embryo and alevin survival as well as enzyme activity in populations that are specifically adapted to warm or cool environments. Thus, small changes in temperature may prove ecologically damaging even though such changes would produce no evidence of acute or chronic physiological stress. Other characteristics, including body morphology, agonistic and rheotactic behavior, and the timing of smolt and adult migrations, are tied to streamflow. Changes in the timing or magnitude of flows because of hydroelectric operations, agricultural diversions, or disruption of hydrologic processes from forest and range practices may effect these characteristics of fish. In this context, the ability of species-specific (versus stock-specific) criteria for water quality, instream flows, and other habitat attributes to adequately protect individual salmonid stocks should be re-evaluated. These stock differences are one reason that hatcheries threaten biological diversity of wild stocks (see Section 6.12).

4.2.4 Mefapopulation Dynamics

The stock concept focuses on the reproductive isolation and subsequent adaptation of local populations to the particular environments that they inhabit. Metapopulation theory is concerned with the behavior of groups of populations, or "metapopulations," that interact via individuals moving among populations through the processes of dispersal or straying (Hanski and Gilpin 1991). The term "metapopulation dynamics" thus describes the long-term behavior of a metapopulation over time.

Early theoretical work on metapopulations focused on extinction and recolonization rates of subpopulations making up a metapopulation (Levins 1969). Local populations within a metapopulation periodically go extinct as a result of natural disturbances or fluctuations in environmental conditions, leaving vacant habitat patches that may subsequently be recolonized by individuals from other populations (Hanski and Gilpin 1991). Under the model of Levins (1969), each subpopulation within the metapopulation has an equal probability of extinction; thus metapopulation persistence requires that, among local populations, the recolonization rate must exceed the extinction rate (Hanski 1991).

More recently, metapopulation models have been proposed that assume various subpopulations play different roles in metapopulation dynamics (Harrison 1991; Hanski 1991). One such model, the core-satellite model, describes a metapopulation where a larger core population gives rise through dispersal to numerous satellite populations (Harrison 1991). In these circumstances, metapopulation persistence depends on the existence of a few extinction-resistant source populations serving as sources of colonists for extinction-prone satellite populations. In a dynamic environment, the role of various subpopulations may change through time; source populations may become sinks and vice versa. Even where local extinction does not occur, depression of populations may influence genetic interactions among populations constituting the metapopulation.

While discussion of metapopulation dynamics of anadromous and resident salmonids is largely absent from the literature (but see Li et al. 1995); a number of principles from metapopulation theory relate to salmonid conservation. Evidence from other

Table 4-4. Examples of local variation in traits of salmonids and their presumed adaptive advantages.*

Trait	Adaptive Advantage	Species (Life cycle phase)†	Source
MORPHOLOGY AND MERISTICS			
Streamlined body and larger fins	Streamlined body and larger fin size adapted for higher water velocities.	Coho salmon (J) Atlantic salmon (J) Pink salmon (A) Chum salmon (A)	Taylor and McPhail (1985) Riddell and Leggett (1981) Beacham (1985); Beacham et al. (1988b) Beacham (1984); Beacham and Murray (1987); Beacham et al. (1988a)
Jaw size and shape	Adapted to local differences in food particle size.	Arctic char	Barbour (1984) Skulason et al. (1989)
Gill raker number and length	Adapted to local differences in food particle size.	Lake whitefish	Ihssen et al. (1981) Lindsey (1981)
BEHAVIOR			
Direction of fry migration	Emerging fry migrate in direction of rearing lakes.	Sockeye salmon (F) Rainbow trout (F)	Brannon (1972); Raleigh (1971) Raleigh (1971); Kelso et al. (1981)
Compass orientation *of* emerging fry	Local differences in orientation facilitate migration to feeding areas.	Sockeye salmon (F)	Quinn (1982, 1985)
Rheotactic behavior	Adapted to local differences in optimal timing of downstream migration.	Chinook salmon (S)	Taylor (1990b)
Homing accuracy	Differences in straying rates potentially reflect differences in environmental stability.	Pink salmon (A)	Bams (1976)
Migratory behavior	Adapted to local differences in predation pressure, local feeding areas, or hydrologic characteristics.	Brown trout (A)	Svardson and Fagerstrom (1982)
Migration timing	Differences between fall and spring races reflect seasonal variation in accessibility to spawning streams.	Chinook salmon (A)	Belding and Kitson (1934); Smith (1969)
Spawning timing	Differences in spawning timing reflect temperature differences in streams.	Pink salmon (A) Chinook salmon (A) Sockeye salmon (A)	Sheridan (1962); Royce (1962) Burger et al. (1985) Brannon (1987)
Agonistic behavior	Reduced level of agonistic behavior in "ocean type" juveniles compared with "stream types" that establish territories.	Chinook salmon (J)	Taylor (1988, 1990b)
	Lower levels of agonistic behavior in populations with high predation; displays may increase risk.	Coho salmon (J)	Rosenau and McPhail (1987)
	Lower levels of agonistic behavior for fish in lakes or other slow\moving habitats. Higher levels in stream-dwelling fish, where territorial defense is advantageous.	Coho salmon (J)	Grant and Noakes (1988) Swain and Holtby (1989)

Table **4-4.** Examples of local variation in traits of salmonids and their presumed adaptive advantages.*

Trait	Adaptive Advantage	Species (Life cycle phase)†	Source
DEVELOPMENTAL BIOLOGY			
Embryo/alevin survival	Survival of embryos/alevins from populations native to coldwater environments greater at low temperatures than for populations from warmwater environments (and vice versa).	Chum salmon (E) Sockeye salmon (E) Pink salmon (E)	Tallman (1986) Beacham and Murray (1989) Beacham (1988); Beacham and Murray (1988)
Developmental rate	Faster development in late spawning stocks may facilitate synchronous emergence with fry of early spawners. Synchrony adaptive for predator swamping or narrow window of favorable oceanic conditions.	Chum salmon (E)	Tallman (1986)
BIOCHEMICAL TRAITS			
Lactate dehydrogenase	Temperature-dependent selection of certain allozymes that are more active at colder or warmer temperatures. Allozymes dominant in northern populations are more active in cold water.	Sockeye salmon	Kirpichnikov and Ivanova (1977)
Esterase-2 locus		Pink salmon	Kirpichnikov and Ivanova (1977)
Isocitrate dehydrogenase		Arctic char	Nyman and Shaw (1971)
Malic enzyme-2 locus		Steelhead trout	Redding and Schreck (1979)
Lactate dehydrogenase 5		Atlantic salmon	Verspoor and Jordan (1989)
		Brown trout	Henry and Ferguson (1985)
PHYSIOLOGICAL TRAITS			
Swim bladder function	Greater swim bladder gas retention ability in fish inhabiting deeper lakes.	Lake trout	Ihssen and Tait (1974)
Swimming ability	Superior prolonged swimming ability in stocks with long freshwater migrations.	Steelhead trout Coho salmon	Taylor and McPhail (1985)
Temperature tolerance	Resistance of fish naturalized to warm water environments and to high temperatures.	Rainbow trout	Morrissy (1973)
Time to smolting	More rapid development adapted to streams with short growing seasons.	Atlantic salmon (S)	Jensen and Johnsen (1986)
DISEASE RESISTANCE			
Resistance to *Cerafornyxa shasta*	Populations that have coevolved with *C. Shasta* have greater resistance than those that have not.	Chinook salmon Coho salmon Steelhead trout	Zinn et al. (1977) Hemmingsen et al. (1986) Buchanan et al. (1983)

Table 4-4. Examples of local variation in traits of salmonids and their presumed adaptive advantages.*

Trait	Adaptive Advantage	Species (Life cycle phase)†	Source
LIFE HISTORY TRAITS			
Large size	Larger size of adults adaptive in populations undertaking difficult migrations or experiencing high flows during spawning.	Chinook salmon Sockeye salmon Chum salmon Brown trout Atlantic salmon Pink salmon	Taylor (1990a) Rogers (1987) Beacham and Murray (1987) L'Abee-Lund et al. (1989) Power (1986) Rogers (1987); Beacham and Murray (1988)
Small size	Adaptation to streams with low summer flows.	Brown trout **(J)**	Borgstrom and Heggenes (1988); Titus and Mosegaard (1989)
Precocious males/parr maturation	Increased incidence of precocious males or parr maturation may be adaptive in populations with long, difficult migrations.	Chinook salmon Atlantic salmon	Myers et al. (1986) Taylor (1989)
Egg size	Late spawners tend to produce smaller faster-developing eggs than early spawners: facilitates synchronous emergence.	Chum salmon	Beacham and Murray (1987) Fleming and Gross (1990)
Fecundity	High fecundity favored in populations that experience high pre-reproductive mortality.	Chinook salmon **(A)**	Healey and Heard (1984)

* Examples are from a review by Taylor (1991).
† Life cycle phases: E = embryo/alevin, F = fry, J =juvenile, S = smolt. A = adult

taxonomic groups suggests that the probability of a local extinction increases with decreasing population size, decreasing size of habitat patches, and increasing isolation from other conspecific populations (reviewed in Hanski 1991; Sjogren 1991). The risk of extinction is also believed to be greater for populations that undergo large natural fluctuations in abundance (Harrison 1991). Recolonization rates are similarly influenced by population size and distance between habitat patches. Re-establishment of populations depends on sufficient numbers of individuals invading that habitat, which in turn depends on dispersal rates, the population size of source populations, the proximity and size of nearby habitat patches, and the availability of suitable migration corridors between patches.

Salmonid metapopulations exhibit many characteristics that would appear to make them vulnerable to extinction. Nehlsen et al. (1991) identified 101 stocks of anadromous salmonids that have had escapements under 200 within the last 1 to 5 years. These stocks are at increased risk of extinction from stochastic genetic, demographic, or environmental events. Many extant salmonid stocks have been eliminated from lower-elevation stream reaches and persist only as remnant populations confined to smaller headwater streams that have been less affected by habitat alterations. First- and second-order streams in steep headwaters tend to be hydrologically and geomorphically more unstable than larger, low-gradient streams. Thus, salmonids are being restricted to habitats where the likelihood of extinction because of random environmental events is greatest. If salmonid metapopulation structure historically resembled the core-satellite model, important source populations may already have been lost, leaving primarily extinction-prone satellite populations, Increased fragmentation of aquatic habitats and isolation of salmon populations reduces the chances that straying individuals from other populations can help restore depleted stocks. Snake River sockeye salmon provide a good example of an isolated population that is unlikely to be rescued by strays from other populations, since the nearest

sockeye stock is over 900 km away (Waples 1995) and straying rates are low. Lastly, salmonids have historically experienced wide interannual variation in numbers as a consequence of variation in both freshwater and marine conditions. Numbers of coho salmon returning to streams in Oregon, Washington, and California can vary by an order of magnitude or more in different years (Hall and Knight 1981). Similar variability in escapement of pink and sockeye salmon has also been documented (Burgner 1991; Heard 1991). The probability of extinction because of fluctuating numbers combined with random environmental events may be particularly high for those species such as pink and coho salmon that have comparatively rigid life histories. In these species, the loss of a particular year-class may have longer-lasting effects than in populations with greater diversity in the age of spawning adults.

A final aspect of metapopulation theory that is relevant to salmonid conservation relates to temporal difference in the dynamics of the local populations that constitute the metapopulation. Hanski (1991) proposed that metapopulation persistence should be greatest where local populations fluctuate independently of each other, i.e., asynchronously, and lowest where local populations fluctuate synchronously in response to regional environmental conditions. The widespread declines in salmon populations throughout the Pacific Northwest suggest that fluctuations in these populations are synchronous, therefore, the risk of metapopulation extinction is relatively high.

4.2.5 Evolutionarily Significant Units

Under the Endangered Species Act or ESA (as amended in 1978), a "species" is defined to include "any subspecies of fish or wildlife or plants, or any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature." For anadromous Pacific salmon and trout, most stocks are, to varying degrees, reproductively isolated—and hence potentially distinct population segments—but **ESA** provides no direction for determining what constitutes a distinct population segment (Waples 1995). To address this concern, the National Marine Fisheries Service (NMFS) has adopted a definition of "species" that is based on the concept of "evolutionarily significant units" or "ESUs" (Waples 1991b). A population is considered an ESU if it meets two criteria: 1) it is substantially reproductively isolated from other conspecific population units, and 2) it represents an important component in the evolutionary legacy of the species (Waples 1991b). For the first criterion to be met, isolation of the population need not be complete, but it must be sufficient to allow accrual of differences in specific traits among populations. Nor is isolation by itself sufficient for a population to be considered distinct. A population may meet the second criterion if it contributes to the overall genetic diversity of the species. In addition, because ecological diversity may foster local adaptations, stocks occupying distinct or unusual habitats or that are otherwise ecologically distinct may also be **ESUs** (Waples 1991b).

The intent of the **ESU** framework is to conserve the genetic diversity of species and the ecosystems that species inhabit, two fundamental goals of ESA (Waples 1991b). The genetic variability within a stock or population represents both the legacy of past evolutionary events and the ability of the population to respond to future environmental changes. The loss of individual stocks or the alteration of the genetic composition of stocks through hatchery introductions can fundamentally alter the ability of the species to cope with local environmental conditions, to respond to environmental change, and hence to persist over the long term.

Waples (1991b) advocates a two-step approach for determining whether a population represents a distinct unit. The first step is to evaluate the degree of reproductive isolation of the population. With salmonids, and particularly anadromous forms, reproductive isolation is rarely complete because of straying and is more a matter of degree. Waples (1991b) recommends several approaches for assessing the degree of reproductive isolation including 1) use of tags to estimate straying rates, 2) intentional genetic marking of populations, **3)** use of genetic indices to estimate levels of gene flow, 4) observation of recolonization rates, and 5) identification of physical or geographic features likely to act **as** barriers to migration. The second step is to evaluate whether the population exhibits evidence of substantial ecological or genetic diversity. Factors to consider include 1) genetic traits, including unique alleles, different allelic frequencies, total genetic diversity; 2) phenotypic traits, including morphological or meristic characters, occurrence of parasites, and disease or parasite resistance; 3) life-history traits, such as time, age, or size at spawning, fecundity, migration patterns, and timing of emergence and outmigration; and 4) habitat characteristics, including temperature, rainfall, streamflow, water chemistry, or biological attributes of the particular system (Waples 1991b).

As Waples (1991b) notes, interpreting data for reproductive isolation is not always straightforward. For example, assessments of straying rates may be confounded by behavior of migratory adults (e.g., temporary entry of fish into non-natal streams). Measures of gene flow may require assumptions of selective neutrality for the alleles used. Assessment of allelic frequencies or presence of unique alleles may be influenced by sampling design, including

number of samples and time of sampling. Similarly, interpreting ecological or genetic diversity data may be equally difficult. Variation in phenotypic and life-history characteristics may be attributable to both genetic and environmental factors; thus phenotypic or life-history variation alone is insufficient for determining population distinctness. The ability to distinguish distinct and unusual habitats is limited by both science and differences of opinion as to what are important habitat characteristics.

Identification of evolutionarily important biological units for protection is further complicated by the fact that a significant number of salmonid stocks have already been lost, and as a result, our understanding of metapopulation structure and function is incomplete. Li et al. (1995) note that few high-quality habitats remain and that many of these lie at the extremes of species' ranges. They argue that conservation strategies should differ depending on metapopulation structure. For example, the classical metapopulation model (Levins 1969) assumes that populations within each metapopulation each carry equal "evolutionary weight," whereas the "core-satellite" model proposes that "core" populations are critical for maintaining smaller satellite populations that might otherwise go extinct. With the classical metapopulation model, the best conservation strategy might be to treat all populations as equally important, protecting as many unique populations as possible in order to protect diversity. In the core-satellite system, emphasis should be placed on protecting core populations, since failure to do so would result in marginal populations of narrow specialists occupying the extremes of the species' range (Li et al. 1995). Waples (1991b) similarly argues that threatened and endangered status should be considered for metapopulations as well as more discrete population units.

Finally, an assumption of the ESU concept is that not all populations need to be protected in order to preserve the genetic integrity of the species (Waples 1991b). Local populations that are not reproductively isolated or that are isolated but fail to exhibit any important and distinctive genetic or life-history traits do not qualify for protection under ESA. In practice, such populations are typically not genetically differentiable from hatchery populations. Where such populations are lost, their ecological function in the aquatic community will also be lost and other organisms may be affected over the evolutionary short term. However, over longer evolutionary time scales the ESU conservation strategy will result in available habitats repopulated by native fish from either within the local ESU or from neighboring ESUs. This should result in fish populations locally adapted or more able to survive and reproduce in the wild, thereby fulfilling their role in the ecosystem.

4.3 Community Level

The biotic communities of aquatic systems are highly complex entities. Within communities, assemblages and species have varying levels of interaction with one another. Direct interactions may occur in the form of predator-prey, competitor, and disease- or parasite-host relationships. In addition, many indirect interactions may also occur between species. For example, predation of one species upon another may enhance the ability of a third species to persist in the community by releasing it from predatory or competitive constraints. These interactions continually change in response to shifting environmental and biotic conditions. Human activities that modify either the environment, the frequency and intensity of disturbance, or species composition can shift the competitive balance between species, alter predatory interactions, and change disease susceptibility, all of which may result in community reorganization.

The role of disturbance in regulating stream-community organization has been a principal focus of aquatic ecology in the past decade. In a recent review, Resh et al. (1988) identify three theories (equilibrium, intermediate disturbance hypothesis, dynamic equilibrium model) that reflect our present understanding of disturbance theory as it relates to stream-community structure. The equilibrium theory proposes that environments are more or less constant and that community organization is determined by biotic interactions, including competition, mutualism, and trophic interactions. The intermediate disturbance hypothesis assumes a competitive hierarchy of species. In the absence of disturbance, superior competitors eliminate inferior ones, whereas in systems with frequent or severe disturbances, resident competitors are eliminated and colonizing species dominate. In systems with intermediate disturbance regimes, species richness is maximized; colonizers exploit disturbed areas and are thus able to coexist with superior competitors. The dynamic equilibrium model proposes that community structure is a function of growth rates, rates of competitive exclusion, and frequency of population reductions. Inferior competitors persist in the community if disturbances occur often enough to eliminate competitive exclusion; however, if disturbances are too frequent, species with long life cycles are eliminated. Species diversity is determined by the influence of the environment on the net outcome of species interactions. Both the intermediate disturbance hypothesis and dynamic equilibrium model emphasize the role of the environment in regulating stream communities, and Resh et al. (1988) conclude that these hypotheses are more generally applicable to stream ecosystems than the equilibrium model. All three models may be

applicable depending on spatial and temporal scales and the type of aquatic system (streams, lakes, reservoirs, and estuaries).

4.3.1 Food Webs

The food energy available to fish and other organisms in aquatic ecosystems comes from two sources: aquatic plants (macrophytes, benthic algae, and phytoplankton) that convert solar energy into biomass and organic litter that falls into the stream and provides the energy base for fungi and bacteria (OWRRI 1995). The relative importance of these energy sources changes with the size and morphology of a river, estuary, or lake system (see Section 4.4.2) and the availability of nutrients in the catchment. Herbivorous aquatic invertebrates consume algae and other aquatic plants, whereas detritivorous invertebrates consume decaying organic matter. Many invertebrates select food on the basis of size, rather than source, while others are generalized feeders. Predatory invertebrates may add an additional trophic level to the food web. Collectively, these invertebrates form an important food base for many juvenile anadromous salmonids and adult trout although some species may feed on other fishes and terrestrial insects that fall into the stream. Fishes, in turn, are consumed by a host of terrestrial and aquatic predators, including other fishes, birds, mammals, reptiles, and amphibians. The resulting food webs can be highly complex, consisting of many species representing several trophic levels.

Food webs may be highly modified by environmental changes, including alterations of the food base; changes in streamflow, temperature, and substrate; and the introduction of non-native organisms. Alterations of individual components of a food web can propagate throughout the system, leading to community wide adjustments in food web composition. For example, impoundments on the Columbia River have shifted the food base from coarse detrital material derived from wetland emergent vegetation and fine material derived from periphyton to a phytoplankton-derived micro-detritus food base, creating numerous adjustments throughout food web (Simenstad et al. 1990; Palmisano et al. 1993b). In the estuary, amphipods and isopods—the preferred food items of salmonid smolts (Dawley 1986)—have now been replaced by suspension feeding epibenthos (Simenstad et al. 1990), which are a primary food source for juvenile American shad. An increasingly favorable environment for shad, coupled with relatively low predation rates, **has** allowed the population to increase dramatically over the last few decades from less than 200,000 to approximately 4 million (Palmisano et al. 1993b). Elimination of woody riparian vegetation from rangeland streams has shifted the food base from coarse, terrestrially derived material to periphyton. The latter is most efficiently consumed by shell-cased macroinvertebrates that are unsuitable prey for juvenile salmonids (Tait et al. 1994).

Changes in water temperatures may change the composition of algal assemblages (Bush et al. 1974); disrupt the development and life-history patterns of benthic macroinvertebrates (Nebeker 1971; Lehmkuhl 1972) and zooplankton (Hutchinson 1967); and decrease the abundance of certain benthic invertebrates, especially species that are stenotherms (Hynes 1970).

Introductions of non-native fish, either as game fish or forage for other fish, have led to food web alterations in most river systems of the Pacific Northwest. In California and Oregon, introduced fishes constitute 35% and 29% of the total species, respectively. The impact of these fish on native species is poorly known, but they are potential predators and competitors of both the juvenile and adult salmonids. The mainstem Columbia river is host to numerous non-native fish, many of them piscivorous, that have acclimated to the lentic habitat of the reservoirs and now dominate many of the trophic pathways. Several mechanisms have been identified that allow introduced fish to succeed in displacing native species, including competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization (Moyle et al. 1986). Similarly, introductions of invertebrates can modify food webs. The introduction of opossum shrimp to Flathead Lake, Montana, resulted in the disappearance of two cladoceran species, which in turn had negative effects on the kokanee salmon that were intended to benefit from the introduction (Spencer et al. 1991).

4.3.2 Competition

Competition among organisms occurs when two or more organismic units (i.e., individuals or species) use the same resources and when availability of those resources is limited (Pianka 1978). Two types of competition are generally recognized: interference competition, where one organism directly prevents another from using a resource through aggressive behavior, and exploitation competition, where one species affects another by utilizing a resource more efficiently (Moyle et al. 1986). Although competition is difficult to demonstrate (Fausch 1988), salmonids likely compete for food and space resources both within species (intraspecific) and between species (interspecific). Within species, stream-dwelling salmonids frequently form dominance hierarchies, with dominant individuals defending holding positions against subordinate fish through agonistic encounters. Evidence suggests that dominant individuals occupy the most energetically profitable holding positions,

which in turn leads to higher growth rates (Fausch 1988). Similar interactions occur between salmonid species; however, in assemblages that have co-evolved, species with similar ecological requirements may segregate according to both micro- and macrohabitats at various life stages.

Changes in physical (e.g., temperature, streamflow, habitat structure) and biological (e.g., food availability, species composition) characteristics of streams and lakes can alter competitive interactions within and among species, potentially resulting in a restructuring of fish communities. In a laboratory study, Reeves et al. (1987) found that stream temperature affected interspecific interactions between juvenile steelhead trout and redside shiner *(Richardsonius* balleatus)—with trout competing more effectively at cold temperatures through interference and shiner competing more successfully at warm temperatures through both exploitation and interference. Cunjak and Green (1986) found that interactions between brook trout *(Salvelinus fontinalis)* and rainbow trout are also influenced by water temperature, rainbow trout being superior competitors at 16°C and brook trout at 9°C. Ratliff (1992) suggests that the decline of bull trout populations in Oregon may in part reflect the inferior ability of bull trout to compete with rainbow, brook, and brown trout *(Salmo trutta)* at warmer temperatures.

Changes in streamflow in the Columbia River system have resulted in increased plankton production, which has apparently increased the success of American shad. Palmisano et al. (1993a, 1993b) conclude that increased numbers of shad, which also feed on benthic invertebrates, may be competing with juvenile salmonids. Cunjak and Green (1984) reported that brook trout tended to dominate social interactions with rainbow trout when in pool habitats, but not in faster waters. Larson et al. (1995) suggest that the dynamics of brook trout and rainbow trout interactions in a southern Appalachian stream may be affected by both temperature and flow conditions. During years of low discharge, rainbow trout encroached on upstream habitats of brook trout possibly because warmer temperatures favored rainbow trout. During periods of higher discharge, encroachment was reversed, presumably because brook trout are better adapted to the steep stair-stepped channel morphology. In general, decreases in streamflow decrease available habitat and may thereby intensify inter- and intraspecific competition for suitable rearing, feeding, spawning, and refuge habitats.

The introduction of non-native species increases the potential for competition in aquatic systems. In natural fish assemblages, salmonids have presumably adapted to other native species with similar ecological requirements through resource partitioning or segregation in time or space. With the introduction of non-native species, however, there has been no opportunity for natural selection to ameliorate competition (Fausch 1988). Several studies have documented influences of non-native species on native salmonids. In a British Columbia lake, cutthroat trout were found to shift from midwater areas when allopatric to littoral zones when sympatric with rainbow trout (Nilsson and Northcote 1981). Dambacher et al. (1992) found that non-native brook trout outcompeted bull trout in Sun Creek, Oregon, in areas of co-occurrence. Intraspecific interactions may also become more intense with the introduction of hatchery fish. Nickelson et al. (1986) concluded that competition between larger hatchery coho salmon and wild juveniles resulted in 44% replacement of the wild fish.

4.3.3 Predation

Adult and juvenile salmonids have evolved strategies to coexist with numerous natural predators including a variety of fish, birds, and mammals. Native fish piscivores include sculpin *(Cottus* spp.), bull trout, rainbow trout, cutthroat trout, northern squawfish *(Ptychochilus oregonensis)*, and possibly white sturgeon *(Acipenser transmountanus)*. These fish prey on juvenile salmonids during instream rearing and during smolt migrations. Northern squawfish are considered important predators of outmigrant salmon and steelhead trout particularly in slackwater habitats (Poe et al. 1991). Bird predators of smolts and juveniles (Kaczynski and Palmisano 1993) include ring-billed gulls *(Larus delawarensis)*, common mergansers *(Mergus merganser)*, herons *(Ardea* spp.), and kingfishers *(Megaceryle alcyon)*. Kingfishers were found to have increased feeding efficiency in slower moving waters. Pinnipeds, including harbor seals *(Phoca vitulina)*, California sea lions *(Zalophus californianus)*, and Stellar sea lions *(Eumetopia jubatus)* are the primary marine mammals preying on salmonids, although Pacific striped dolphin *(Lagenorhynchus obliquidens)* and killer whale *(Orcinus orca)* may also prey on adult salmonids. Seal and sea lion predation is primarily in saltwater and estuarine environments though they are known to travel well into the freshwater environment after migrating fish. All of these predators are opportunists, searching out locations where juveniles and adults are most vulnerable.

Habitat alterations can affect predation rates by reducing cover, which increases vulnerability to capture by predators; altering flow regime and water velocity, which may favor certain piscivorous fishes; modifying temperature, which affects the metabolism of piscivorous fish and the ability of fish to elude predators; and by obstructing passage, which may

delay migrations and thereby increase exposure to predators. In the Columbia Basin altered flow regimes have contributed to the increased success of northern squawfish, walleye *(Stizostedion vitreum),* and smallmouth bass (*Micropterus dolomieu*), which tend to avoid areas of high water velocity (Faler et al. 1988). Laboratory experiments with squawfish have shown that maximum consumption of salmonid smolts increased from 0.5 smolts per day at 8.3°C to 7 smolts per day at 21.7°C (Vigg and Burley 1991), indicating that temperature increases may indirectly cause greater predation on juvenile salmonids (Palmisano et al. 1993b). The high incidence of predation by sea lions at such places as Ballard Locks in Washington is in part attributable to the unnatural congregations of fish as they attempt to pass through the locks.

4.3.4 Disease and Parasitism

Salmonids are affected by a variety of bacterial, viral, fungal, and microparasitic pathogens. In the Pacific Northwest, numerous diseases may result from pathogens that occur naturally in the wild or that may be transmitted to wild fish via infected hatchery fish. Among these are bacterial diseases, including bacterial kidney disease (BKD), columnaris, furunculosis, redmouth disease, and coldwater disease; virally induced diseases, including infectious hepatopoietic necrosis (IHN) and erythrocytic inclusion body syndrome (EIBS); protozoan-caused diseases, including ceratomyxosis and dermocystidium; and fungal infections, such as saprolegnia (Bevan et al. 1994a). Brief descriptions of the most prevalent pathogens and the associated diseases are shown in Table 4-5.

Water temperature greatly influences the immune system of fishes, the number and virulence of pathogens, and in the case of microparasites, the occurrence of infective life stages in natural and aquacultural environments. Consequently, changes in water temperatures caused by forest and range practices, dams, and irrigation can alter the susceptibility of salmonids to infection by these pathogens. Most work on fish pathogens has concerned fish in culture situations, and the incidence of disease and its role of fish population dynamics and in structuring fish assemblages in natural waters is poorly understood (Austin and Austin 1993). Nevertheless, laboratory studies indicate that water temperature has a direct effect on the infection rate of most pathogens and the mortality rate of infected salmonids. With most pathogens, the susceptibility of salmonids to infection tends to increase with increasing water temperatures, although mortality from coldwater disease is greater when temperatures are lower (Holt et al. 1993). A summary of the general relationship between temperature and important pathogens in Pacific Northwest environments is shown in Table 4-5.

Several recent epizootics indicate that temperature may play a significant role in mediating disease in natural populations. Prespawning mortality in fall chinook salmon was highly correlated with mean maximum stream temperatures in the Rogue River (Oregon) during August and September, with mortality rates increasing abruptly at temperatures greater than 20°C (ODFW 1992). *Flexibacter colurnnaris* was commonly found in dead and dying fish and was presumed to be the primary agent causing mortality. Release of warm reservoir water during the late summer and early fall has been implicated in outbreaks of *Dermocystidium salmonis* in anadromous fish in the lower Elwha River, Washington (NPS et al. 1994). In 1992, approximately two-thirds of the adult chinook population in the lower river died prior to spawning (Wunderlich et al. 1994).

While epizootics provide the most dramatic examples of the potential for pathogens to affect salmonid populations, sublethal chronic infections can impair the ability of fish to perform in the wild and thereby contribute secondarily to mortality or reduced reproductive success. Fish weakened by disease are more sensitive to other environmental stresses. Furthermore, infected fish may become more vulnerable to predation (Hoffman and Bauer 1971), or less able to compete with other species. For example, Reeves et al. (1987) found that the interspecific interactions between juvenile steelhead trout and redside shiner were affected by water temperature. They speculated that these differences were in part because most juvenile steelhead were infected with *F. columnaris* at high temperatures, whereas shiners showed a higher incidence of infection at lower temperatures.

The susceptibility of salmonids to disease may be affected by other stressors, including dissolved oxygen, chemical pollution, and population density. Temperature may interact synergistically with these factors, causing disease to appear in organisms that might be resistant in the absence of other forms of stress. Susceptibility also varies among salmonid species and life stages. For example, older chinook have been shown to be more resistant to ***F.*** *colurnnaris* than younger fish (Becker and Fujihara 1978).

4.4 Connectivity Among Processes

The biotic communities found in streams and rivers reflect physical and chemical gradients that occur both across the landscape and along a stream from the headwaters to the ocean. In the preceding sections, we have reviewed fundamental biological processes that occur at the level of organisms,

Table 4-5. Pathogens of salmonids found in Pacific Northwest waters.

Pathogen	Disease	Comments
BACTERIA		
Aeromonas salmonicida	Furunculosis	Obligate pathogen of fish. Low mortality at temperatures < 6.7°C. Increasing mortality at 9.4°C. At 20.5°C, 93%–100% mortality for all species. (Groberg et al. 1978)
Aeromonas hydrophila		Stress facilitated infection. Mortality is associated with elevated water temperatures (> 9.4°C), presence of pollutants (particularly nitrate at ≥ 6 mg/L). (Austin and Austin 1993)
Flexibacter columnaris	Columnaris	Low mortality at temperatures < 15°C. Increasing mortality at 20°C for all species. Virulence at low temperatures depends on specific strain. Naturally occurring bacteria present at low levels in resident fish (suckers, carp, and whitefish). Stress increases fish susceptibility. High density increases potential for contact. (Inglis et al. 1993).
Flexibacfer psychrophilus	Coldwater disease	Appears in spring when temperatures are between 4–10°C; 30%–50% mortality for infected alevins; Quickest mortality at 15°C. Mean time to death increases with temperatures from 15–23°C. Mode of transmission unknown. Resident salmonids are probable carriers. Possible vertical transmission. (Inglis et al. 1993)
Renibacfenum salmoninarum	Bacterial kidney disease (BKD)	Obligate pathogen of fish. Disease progresses more rapidly at higher temperatures (15–20.5°C), but mortality may be highest at moderate temperatures (12°C). Transmission is both horizontal and vertical (intraovum). Crowding and diet stress can increase susceptibility. (Inglis et al. 1993; Fryer and Lannan 1993)
Yersinia rucken	Redmouth disease	Mortality may be low in chronic infections but becomes much higher with stress from poor water conditions (elevated temperatures, ammonia, metabolic waste, copper). Transmission through water, via baitfish, introduced fish, bird feces, fish farms. (Inglis et al. 1993)
FUNGI		
Saprolegnia		Ubiquitous in water. Transmitted horizontally or from substratum to fish. Elevated temperatures increase growth rate. If untreated, progressive and terminal. (Bell 1986)

Table 4-5. Pathogens of salmonids found in Pacific Northwest waters.

Pathogen	Disease	Comments
PROTOZOANS		
Ceratomyxa shasta	Ceratamyxosis	Endemic to many river systems of Northwest. Temperature dependent; increasing mortality for fish exposed at temperatures ≥ 10°C. High mortality for nonadapted (no genetic resistance) species and stocks. (Ratliff 1983).
Dermocystidium salmonis		Pathogen of salmonids in Pacific Northwest. Horizontal transmission through water. Uptake is via gills. Epizootics appear to be temperature dependent. (Olson et al. 1991)
Ichthyobodo/Costia spp.		Ectoparasite affects osmoregulation. Juvenile salmonid mortality high (63%–70% in 48 h tests) upon introduction to marine waters. (Urawa 1993)
Myxobolus cerebralis	Whirling disease	Salmonid infection by mature triactinomyxon via ingestion or through gills. Horizontal transmission. Intermediate host is tubifex worm from soft mud habitats. Lethal to salmonids. (Rich Holt, personal communication, 1995).
VIRUSES		
infectious Hepatopoietic Necrosis Virus	IHN	Endemic to most areas. High for young fry. Most mortality occurs at temperatures of 12°C or less. Some outbreaks at 15°C. At temperatures over 10°C, disease produces less mortality but leads to more carriers of disease. (Wolf 1988).
Erythrocytic Inclusion Body Syndrome Virus	EIBS	Potential vertical transmission and known horizontal transmission. Greatest mortality of salmonids found at 8–10°C. (Takahashi et al 1992; Leek 1987)

populations, and communities, and the relationship between these processes and habitat characteristics that are affected by human activities. In this section, we discuss two concepts, the river continuum concept and the ecoregion concept, that address spatial relationships between these physicochemical and biological processes. The river continuum concept (Vannote et al. 1980) focuses on interrelationships between physical and biological processes along streams from their headwaters to the ocean. The ecoregion concept relates regional patterns in physical and chemical gradients to the biological communities contained therein.

4.4.1 River Continuum Concept

The river continuum concept (Vannote et al. 1980) proposes that the physicochemical variables (e.g., light, nutrients, organic materials) within a river system change in a systematic way as a stream flows from headwaters to larger river systems to the ocean, and that the biological communities found along this gradient change accordingly (Figure 4-2). In forested headwater reaches, energy inputs are dominated by coarse allochthonous materials, particularly leaf litter from riparian vegetation. As streams increase in size, canopy cover becomes less complete and more light reaches the stream; consequently, the contribution of instream primary

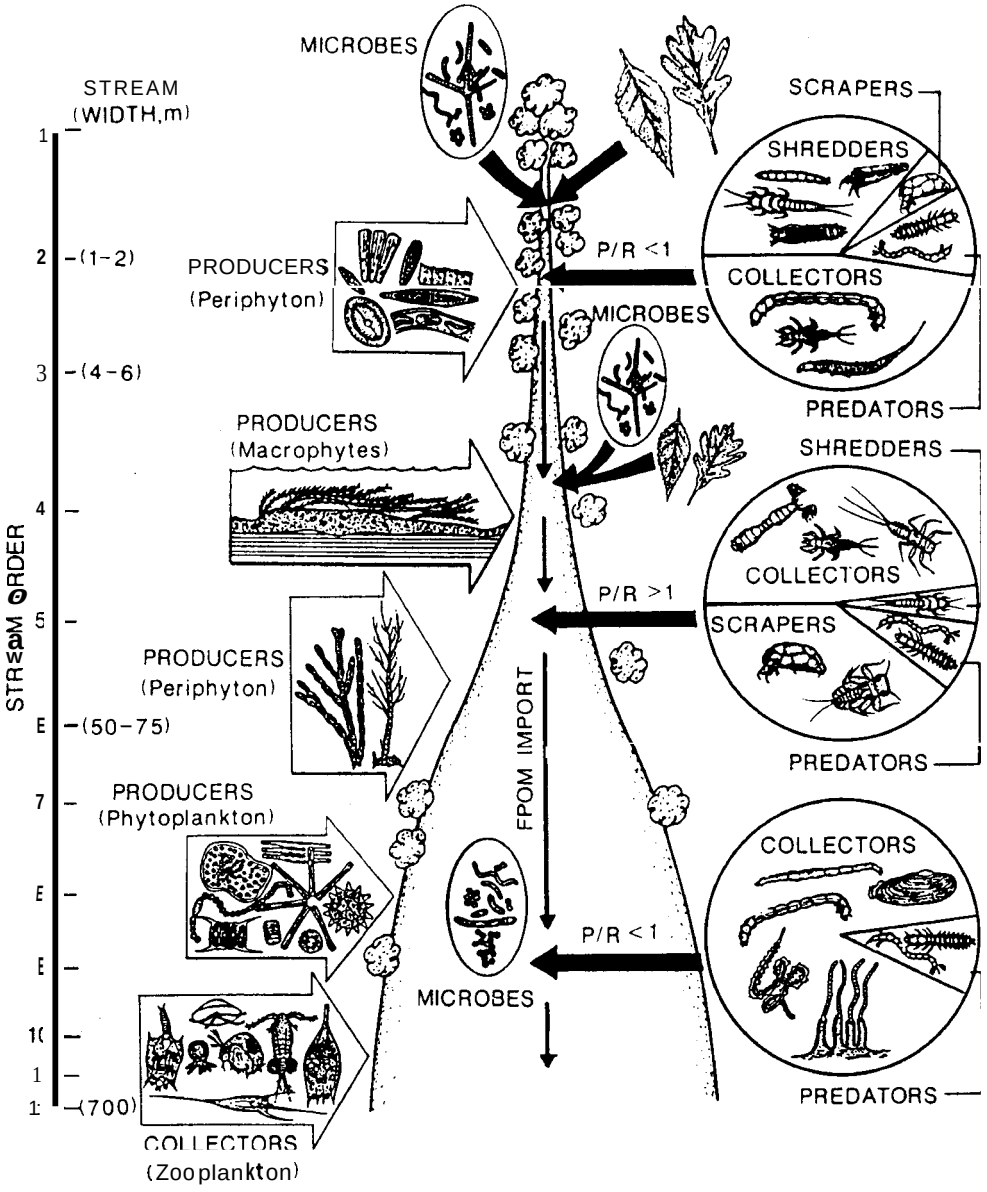


Figure 4-2. Trends in energy sources, ratios of autotrophic production to heterotrophic respiration, and functional groups along a river continuum. From Vannote et al. (1980). Reproduced with permission from the publisher.

production from algae and macrophytes increases relative to energy derived from allochthonous materials. In still larger systems, fine particulate material transported from upstream areas forms the dominant energy source, particularly where depth and turbidity limit algal growth. These gradients lead to corresponding changes in the biological communities that use these changing energy sources. Invertebrate communities shift from those dominated by shredders and collectors in small streams, to collectors and grazers in mid-order streams, to mostly collectors in large rivers. Fish assemblages shift from invertivores in headwater reaches, to piscivores and invertivores in mid-order reaches, and include some planktivores in larger rivers.

Although the river continuum concept was developed in forested biomes, it can also be applied more generally. Meehan (1991) suggests that meadows and deserts, which lack shading and have reduced allochthonous inputs, obtain most of their energy from autochthonous sources, in contrast to woodland streams which have stronger terrestrial influences and therefore greater quantities of coarse particulate detritus. They conclude that desert streams are more similar to the downstream reaches of forested streams. Minshall et al. (1985) illustrate this conceptually by proposing a sliding scale to indicate that streams enter the continuum at different points. Similarly, primary production by algae may be high in headwater streams of alpine systems, where riparian inputs are comparatively low. Consequently these systems may have a different sequence in the biological communities along the continuum.

4.4.2 Ecoregions

Ecological processes that influence salmonids and other aquatic species in the Pacific Northwest vary greatly across the landscape because of the high diversity of climate, topography, geology, vegetation, and soils. Classifications of ecoregions represent attempts to identify areas of relative homogeneity in ecological systems or in the relationships between organisms and their environments (Omernik and Gallant 1986). Several Federal agencies, including the Environmental Protection Agency (Omernik and Gallant 1986; Omernik 1987), the U.S. Forest Service (Bailey 1978), and the Soil Conservation Service (Norris et al. 1991) have developed or are in the process of developing ecoregion classifications in order to address spatial issues in the management of natural resources. Landscapes, water bodies, and the biota that they support are expected to be similar within an ecoregion and to differ between ecoregions. We believe some form of ecoregion classification will be essential to defining the natural range of physical, chemical, and biological characteristics of salmonid habitats across the landscape.

The various processes for delineating ecoregions differ. Omernik and his colleagues synthesized a number of factors (climate, geology, topography, soil, vegetation, land cover) to assess patterns at multiple spatial scales. Bailey considered many of the same factors but used only one at any single scale of resolution. For example, his first divisions were by climatic patterns and his last were by vegetation. The Soil Conservation Service, as might be expected, focused on soil and agricultural land uses. Omernik's approach is favored by many State water quality agencies because of its ability to assess patterns at multiple scales and its adaptability, and it has been recommended by other scientific organizations (SAB 1991; NRC 1992).

Although there are serious limitations to the application of Omernik's ecoregions at the site or small catchment scales, they are useful for stratifying the regional variability of the Pacific Northwest (Table 4-6) into relatively distinct units. In addition, ecoregions offer a framework for aggregating and extrapolating data collected at the local level. A regional perspective is also essential for managing widely distributed resources, such as Pacific salmonids, because of the natural variability among sites and the human tendency to focus on local issues while losing sight of regional ones. In addition, subregions can be developed in a hierarchical manner to facilitate more precise landscape classification at local scales (Clarke et al. 1991; Bryce and Clarke 1996). Direct applications of ecoregion concepts to aquatic ecosystems have demonstrated the utility of this approach. Whittier et al. (1988) showed that fish assemblages in rivers and small streams exhibited patterns concordant with Omernik's ecoregions in Oregon. In evaluating a number of different data sets from basin to State scales, Hughes et al. (1994) found ecoregions that differed markedly supported dissimilar fish assemblages, similar ecoregions supported more similar fish assemblages, and within-region variation was less than among-region variation.

4.5 Summary

In the preceding sections, we have discussed biological processes at three levels of biological organization: organisms, populations, and communities. Grouping processes into these discrete categories serves to simplify thinking about the effects of environmental perturbations on salmonids and their ecosystems, but it should be reiterated that salmonids are simultaneously affected by processes occurring at all levels of biological organization. Physiological stresses influence the ability of salmonids to acquire food and defend space from competitors, to escape or avoid predators, and to fend off infectious diseases and parasites, all of

Table **4-6.** Predominant characteristics of ecoregions in the Pacific Northwest. From Omernik and Gallant (1986).

Ecoregion	Land surface form	Potential natural vegetation	Land use	Soils*
Coast Range	Low to high mountains	Spruce/cedar/hemlock, cedar/hemlock/Douglas-fir. redwood	Forest and woodland mostly ungrazed	Udic soils of high rainfall areas
Puget Lowland	Tablelands with moderate relief, plains with hills or mountains, or open hills	Cedar/hemlock/Douglas-fir	Mosaic including forest, woodland, pasture, cropland	Alfisols, Inceptisols, Mollisols, Spodosols, and Vertisols of valleys
Willamette Valley	Plains with hills, or open hills	Cedar/hemlock/Douglas-fir, mosaic of Oregon oakwoods and cedar/ hemlock/Douglas-fir	Primarily cropland with some inter-spersion of pasture, woodland, and forest	Xeric Mollisols, Vertisols, and Alfisols of interior valleys
Cascades	High mountains	Silver fir/Douglas-fir, fir/hemlock, western spruce/fir, Douglas-fir, cedar/hemlock/Douglas-fir, spruce/cedar/ hemlock	Forest and woodland mostly ungrazed	Udic soils of high rainfall mountains
Sierra Nevada	High mountains	Mixed conifer forest (fir, pine, Douglas-fir), red fir, lodgepole pine/ subalpine forest (hemlock)	Forest and woodland grazed	Xeric soils of moderate rainfall areas
Southern and Central California Plains and Hills	Irregular plains, tablelands of moderate to considerable relief, low mountains	California oakwoods, chaparral (manzanita, ceanothus), California steppe (needlegrass)	Open woodland grazed	Light-colored soils of subhumid regions
Central California Valley	Flat plains	California steppe (needlegrass), tule marshes (bulrush, cattails)	Irrigated agriculture, . cropland with grazing land	Recent alluvial soils, light-colored soils of the wet and dry subhumid regions
Eastern Cascades Slopes and Foothills	Varied: tablelands with moderate to high relief, plains with low mountains, open low mountains, high mountains	Western ponderosa pine	Forest and woodland grazed	Xeric soils of moderate rainfall areas
Northern Rockies	High mountains	Cedar/hemlock/pine, western spruce/fir, grand fir/Douglas-fir, Douglas-fir	Forest and woodland mostly ungrazed	Eastern interior mountain soils with acidic rock types, Inceptisols
Columbia Basin	Varied: irregular plains, tablelands with moderate to high relief, open hills (excludes extremes)	Wheatgrass/bluegrass, fescue/wheatgrass, sagebrush steppe (sagebrush, wheatgrass)	Mostly cropland, cropland with grazing land	Xerolls, channeled scablands
Blue Mountains	Low to high open mountains	Grand fir/Douglas-fir, western ponderosa pine, western spruce/fir, Douglas-fir	Forest and woodland grazed	Soils of eastern interior mountains, Mollisols, Inceptisols
Snake River Basin/High Desert	Tablelands with moderate to high relief, plains with hills or low mountains	Sagebrush steppe (sagebrush, wheatgrass), saltbushlgreasewood	Desert shrubland grazed, some irrigated agriculture	Aridisols, aridic Mollisols

* Soils are presented in this table as they appear from mapped units of resource soil maps.

which affect community structure. Populations have evolved specific mechanisms for coping with environmental conditions in their natal and rearing streams. These adaptations include morphological, biochemical, physiological, behavioral, and developmental traits that allow fish to survive and thrive with the specific physical, chemical, or biological constraints imposed by the environment and that ensure specific activities (e.g., timing of migration and emergence) coincide with favorable environmental and ecological conditions. Adaptation is also evident in life-history strategies (e.g., fecundity and straying rates) that accommodate natural disturbance regimes and allow populations to persist over evolutionary time. Unlike the biological diversity of fishes in the Mississippi Basin, which centers on species diversity, the fish diversity in the Pacific Northwest centers on stock and life-history diversity. The evolution of a wide variety of life-history strategies has allowed salmonids to invade and thrive in the diverse habitats of the Pacific Northwest. The linkage between biological communities and the physical and chemical characteristics of streams are illustrated through the concepts of the river continuum and ecoregions, which offer means for assessing patterns in aquatic community structure across the landscape and for predicting the response of aquatic ecosystems to anthropogenic disturbance. These concepts are essential in developing site-specific and region-specific salmonid conservation strategies and goals.

5 Habitat Requirements of Salmonids

Karr (1991) defines biological integrity as "the ability to support and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitat of the region." He further states that a biological system can be considered "ecologically healthy" when "its inherent potential is realized, its condition is stable, its capacity for self-repair when perturbed is preserved, and minimal external support for management is needed."

Specific attributes of streams and lakes, such as streamflow, water temperature, substrate, cover, and dissolved materials—all the elements typically associated with the term habitat—are the result of physical, chemical, and biological processes operating throughout a watershed and across the landscape (see Chapters 3 and 4). Protecting and restoring desirable habitat-attributes of streams and lakes for salmonids requires that the natural processes that produce these characteristics be maintained or restored. If processes are protected, in other words, desirable aquatic-habitat characteristics will develop; if the processes are altered, the integrity of the aquatic ecosystem and its ability to support salmonids are diminished. The Aquatic Conservation Strategy Objectives detailed by FEMAT (1993) directly reflects these concepts.

To assess the habitat requirements of salmonids, four principles need to be considered: 1) all watersheds and streams are different to some degree in terms of their temperature regimes, flow regimes, sedimentation rates, nutrient fluxes, physical structure, and biological components; 2) the fish populations that inhabit a particular body of water have adapted—biochemically, physiologically, morphologically, and behaviorally—to the natural environmental fluctuations that they experience and to the biota with which they share the stream, lake, or estuary (see Section 4.2.3); 3) the specific habitat requirements of salmonids differ among species and life-history types and change with season, life stage, and the presence of other biota: and 4) aquatic ecosystems are changing over evolutionary time. From these general principles, there are obviously no simple definitions of desirable habitat characteristics of salmonids. Defining acceptable or natural ranges of variability for specific habitat attributes is not only difficult, it can be misleading as well. For example, the same total sediment yield in two different watersheds may affect salmonid habitats differently, depending on geology, topography, hydrology, stream size, and the abundance of large woody debris. Similarly, Behnke (1992) has suggested that stocks of trout native to warmer streams may exhibit greater tolerance to high temperature extremes than stocks inhabiting naturally cooler waters; simply defining the range of temperatures at which a species has been observed does not ensure that stocks will be "safe" or healthy as long as temperatures remain in that range. The FEMAT (1993) report concluded that current scientific information is inadequate to allow definition of specific habitat requirements of salmonids throughout their life histories. These points further emphasize the need to maintain the integrity of aquatic ecosystems.

Karr (1991) identified five classes of environmental factors that affect the biotic integrity of aquatic ecosystems—food (energy) source, water quality, habitat structure, flow regimes, and biotic interactions—as well as ecological changes that may occur in response to human-induced alterations of these factors (Figure 5-1). Although this model was intended to address all aquatic biota, the elements provide a useful framework for discussing salmonid habitat requirements. In Section 5.1, we use the model of Karr (1991) to outline general habitat requirements of salmonids, focusing on processes and characteristics that must be maintained in order to ensure the ecological health of aquatic ecosystems. In Section 5.2, we review specific habitat requirements of Pacific salmonids at each life stage: adult migration, spawning and incubation, rearing, and juvenile migration. An extended discussion of water-quality concerns is presented under "general habitat requirements"; it is beyond the scope of this report to comprehensively review the effects of toxic substances on each life stage.

5.1 General Habitat Requirements

Everest et al. (1985) noted that although each species of anadromous salmonid differs somewhat in its specific habitat requirements, all share some common habitat needs. Extending their list to include resident species, all salmonids require sufficient invertebrate organisms for food; cool, flowing waters

Ecological impact of human-induced alterations

1. food (energy) source
- type, amount, and particle size of organic material entering a stream from the riparian zone versus primary production in the stream
- seasonal pattern of available energy

→
- decreased coarse particulate organic matter
- increased fine particulate organic matter
- increased algal production

2. water quality
- temperature
- turbidity
- dissolved oxygen
- nutrients (primarily nitrogen and phosphorous)
- organic and inorganic chemicals, natural and synthetic
- heavy metals and toxic substances
- pH

→
- expanded temperature extremes
- increased turbidity
- altered diurnal cycle of dissolved oxygen
- increased nutrients (especially soluable nitrogen and phosphorous)
- increased suspended solids
- increased toxics
- altered salinity

3. habitat structure
- substrate type
- spawning, nursery, and hiding places
- diversity (pools, riffles, woody debris)
- basin size and shape

→
- decreased stability of substrate and banks due to erosion and sedimentation
- more uniform water depth
- reduced habitat heterogeneity decreased channel sinuosity
- reduced habitat areas due to shortened channel
- decreased instream cover and riparian vegetation

4. flow regime
- water volume
- water depth and current velocity
- temporal distribution of floods and low flows

→
- altered flow extremes (both magnitude and frequency of high and low flows)
- increased maximum flow velocity
- decreased minimum flow velocity reduced diversity of microhabitat velocities
- fewer protected sites

5. biotic interactions
- competition
- predation
- disease
- parasitism

→
- increased frequency of diseased fish altered primary & secondary production
- altered trophic structure
- altered decomposition rates and timing disruption of seasonal rhythms
- shifts in species composition and relative abundances
- shifts in invertebrate functional groups (increased scrapers and decreased shredders) shifts in trophic guilds (increased omnivores and decreased piscivores)
- increased frequency of hybridization
- increased frequency of non-native species

Figure 5-1. Five major classes of environmental factors that affect aquatic biota. Arrows indicate the kinds of effects that can be expected from human activities (modified from Karr 1991).

free of pollutants; high dissolved oxygen concentrations in rearing and incubation habitats; water of low sediment content during the growing season (for visual feeding); clean gravel substrate for reproduction; and unimpeded migratory access to and from spawning and rearing areas.

5.1.1 Food (Energy) Source

As discussed in Section 4.1, salmonids require sufficient energy to meet their basic metabolic needs, to grow, and to reproduce. Maintaining the integrity of aquatic ecosystems depends on maintaining the natural spatial and temporal patterns and amount of primary production. In streams where energy inputs are dominated by allochthonous materials, detrital particles generally are larger than in streams where autochthonous production dominates. In addition, in streams with an intact riparian canopy, the timing and type of material delivered to the channel differs between coniferous and deciduous forests. Together, these factors determine the abundance and species composition of aquatic invertebrates, which are the principal food source for most salmonids. Removal of riparian vegetation in smaller streams changes the dominant energy inputs from allochthonous to autochthonous sources. The conversion of riparian vegetation from conifer-dominated communities to deciduous-dominated communities, or from shrub-dominated to grass-dominated communities, alters the type of food energy available to the system, the temporal patterns of allochthonous inputs, and the invertebrate communities that feed on those resources. Although not all of these changes are necessarily detrimental to salmonids, they represent fundamental changes to ecosystem function. Streams with anadromous fish populations have an additional important source of nutrients in the form of salmon carcasses (see Section 3.8.2), which may contribute substantially to the productivity of the system.

Physical habitat complexity influences the retention and processing of organic materials within streams and rivers. In addition, characteristics of the physical and chemical environment—temperature, streamflow, turbidity, nutrient availability, and physical structure—all influence the composition and abundance of invertebrate communities within streams, lakes, and estuaries, as well as the ability of salmonids to obtain these food resources (see Section 4.1.1). Thus physical and chemical processes must be maintained to ensure that food resources remain within the natural range of abundance for the particular site.

5.1.2 Water Quality

Water temperature, turbidity, dissolved gases (e.g., nitrogen and oxygen), nutrients, heavy metals, inorganic and organic chemicals, and pH all influence water quality and the ability of surface waters to sustain fish populations. With the exception of organic and inorganic chemicals of anthropogenic origin, each of these factors is naturally occurring and exhibits daily or seasonal fluctuations in concentration or magnitude. If the magnitude or concentration of any of these factors exceeds the natural range for a specific location and time of year, biological processes are altered or impaired.

Ternperatlre

Perhaps no other environmental factor has a more pervasive influence on salmonids and other aquatic biota than temperature. The vast majority of aquatic organisms are poikilothermic—their body temperatures and hence metabolic demands are determined by temperature (see Section 4.1). Consequently, virtually all biological and ecological processes are affected by ambient water temperature. Many effects of temperature on these processes have been discussed elsewhere in this document. Below is a brief list of some of the more important physiological and ecological processes affected by temperature, referenced to sections of this document where more detailed discussions may be found

- Decomposition rate of organic materials
- Metabolism of aquatic organisms, including fishes (Sections 4.1.1, 4.1.5)
- Food requirements, appetite, and digestion rates of fishes (Section 4.1.1)
- Growth rates of fish (Section 4.1.1)
- Developmental rates of embryos and alevins (Section 4.1.2)
- Timing of life-history events including adult migrations, fry emergence, and smoltification (Section 4.1.4)
- Competitor and predator-prey interactions (Sections 4.3.2, 4.3.3)
- Disease-host and parasite-host relationships (Section 4.3.4)
- Development rate and life history of aquatic invertebrates

From this list, it is evident that protection and restoration of salmonid habitats requires that temperatures in streams and lakes remain within the natural range for the particular site and season.

Most of the literature on salmonid temperature requirements refers to "preferred", "optimal", or "tolerable" temperatures or temperature ranges (Everest et al. 1985: Bell 1986: Bjornn and Reiser 1991). Preferred or optimal temperatures are generally derived in laboratory studies of behavior (e.g., temperature selection) or performance (e.g., growth, survival, metabolic scope). In general, the term "preferred temperature" is used to describe the

temperature to which, given unlimited acclimation time, a fish will ultimately gravitate towards (Fry 1947). The "optimum temperature" means the temperature at which a fish can best perform a specific activity. The "tolerable temperature" range includes temperatures at which fish can survive indefinitely. Although studies of temperature preferences, optima, and tolerances are useful in establishing general physiological requirements, they do not address the ecological requirements of salmonids or local adaptation to specific thermal regimes. For this reason, water-quality criteria that are designed to prevent temperatures from exceeding physiologically stressful levels alone are unlikely to prevent more subtle ecological changes.

Turbidity and Suspended Solids

Turbidity in streams is caused by phytoplankton and by inorganic and organic materials that become suspended during high flow conditions. Inorganic and organic solids enter the aquatic environment in surface runoff, or as particles derived from erosion associated with natural (e.g., slumping of unstable banks, storm runoff, volcanoes) or anthropogenic activities (e.g., forestry, grazing, mining, and agricultural practices) (Leidy 1980; Stumm and Morgan 1981; Dickson et al. 1987; Adriano 1992; Hem 1992).

Turbidity and suspended solids in surface waters can effect periphyton and phytoplankton by reducing light transmission and by causing physical damage through abrasion and scouring (Chandler 1942; Chapman 1963; Bullard 1965; Cairns et al. 1972). A number of studies have indicated that turbidity is a major factor controlling phytoplankton abundance (Buck 1956; Cordone and Pennoyer 1960; Herbert et al. 1961; Benson and Cowell 1967; Sherk et al. 1976). In addition, diminished light penetration and streambed stability can lead to reductions in algal productivity (Samsel 1973) and changes in plant species composition. Samsel (1973) found that a reduction of transparency of about 50% caused a threefold reduction in algal productivity in a Virginia impoundment. Chapman (1963) noted that moving sediment may grind or dislodge algae. Shifting of deposited sand (0.008–0.015 inches) prohibited establishment of periphyton along an English riverbed (Nuttall 1972).

Siltation reduces the diversity of aquatic insects and other aquatic invertebrates by reducing interstices in the substrate. When fine sediment is deposited on gravel, species diversity and densities drop significantly (Cordone and Pennoyer 1960; Herbert et al. 1961; Bullard 1965; Reed and Elliott 1972; Nuttall and Bielby 1973; Bjornn et al. 1974; Cederholm et al. 1978). Deposited sediment may reduce accessibility to microhabitats by embedding the edges of cobbles (Brusven and Prather 1974), and it may also entomb benthic organisms, which then die of oxygen depletion (Ellis 1931). Suspended sediments also limit benthic invertebrates (Tarzwell 1938; Rees 1959; Branson and Batch 1971). In a 10-year stream survey Roback (1962) found numbers of caddis fly larvae genera decreased from 16 to 7 at sediment concentrations in excess of 500 ppm. Addition of more than 80 ppm of inert solids to the normal suspended particle concentration of 40 ppm caused a 60% reduction in population of riffle macroinvertebrates (Gammon 1970). Estuarine copepods ingested fewer food organisms as silt concentration increased (Sherk et al. 1976).

Siltation and turbidity adversely affect fish at every stage of their life cycle (Iwamoto et al. 1978). In general, deposited sediments have a greater impact on fish than do suspended sediments: spawning and incubation habitats are most directly affected (see Section 5.2.2). Particulate materials physically abrade and mechanically disrupt respiratory structures (e.g., fish gills) or surfaces (e.g., respiratory epithelia of benthic macroinvertebrates) in aquatic vertebrates and invertebrates (Rand and Petrocelli 1985). Sediment covers intergravel crevices which fish use for shelter, thereby decreasing the carrying capacity of streams for young salmon and trout (Cordone and Kelley 1961; Bjornn et al. 1974). Fish vacate pools in summer after heavy accumulation of sediments (Gammon 1970). Finally, turbidity affects light penetration, which in turn affects the reactive distance of juvenile and adult salmonids for food capture (see Section 5.2.2).

Although salmonids typically prefer water with low turbidity and suspended sediment content, low levels of turbidity may have beneficial effects. Particulates and dissolved chemical solids, including materials harmful to salmonids, may adsorb to the surfaces of colloidal materials, which in turn can reduce their bioavailability. Thus, adverse effects potentially associated with exposures to inorganic and organic chemicals may be diminished, and biological processes associated with adsorption of dissolved organic solids (e.g., microbial transformation) may enhance the biodegradation and detoxification of organic chemicals in the water (Dickson et al. 1987; Rand and Petrocelli 1985; Adriano 1992; Hem 1992). While adsorption associated with colloids may attenuate adverse biological effects associated with some chemicals, toxicity of other dissolved chemical solids may increase because of interactions with colloidal materials in the water column. The exposure of fish to heavy metals may increase or the solubilization of heavy metals from otherwise insoluble metal compounds may increase in the presence of suspended solids having a high colloidal

content (Leidy 1980; Rand and Petrocelli 1985; Brown and Sadler 1989).

Dissolved Oxygen and Nitrogen Gases

All salmonids require high levels of dissolved oxygen (DO). Reduced levels of oxygen can affect the growth and development of embryos and alevins, the growth of fry, and the swimming ability of adult and juvenile migrants. In most natural situations, DO levels are sufficient to allow normal function, but concentrations may be reduced by large amounts of organic debris, nutrient enrichment from sewage treatment plants and agricultural runoff, and excessively high temperatures. Bjornn and Reiser (1991) reviewed a number of papers and concluded that while thresholds for survival are generally low (3.3 mg/L), growth and food conversion efficiency are affected at DO levels of 5 mg/L, and that DO levels of 8—9 mg/L or more are needed to ensure that normal physiological functions of salmonids are not impaired. EPA's water-quality criteria for dissolved oxygen are 9.5 mg/L for a 7-day mean and 8.0 mg/L for a l-day minimum (EPA 1986). Supersaturation of oxygen gas may occur associated with spills from dams or highly turbulent waters. The EPA standard for maximum levels of oxygen is 110% of normal saturation. **A** more detailed discussion of specific oxygen requirements at each life stage is presented in Section 5.2.

Numerous studies of nitrogen supersaturation indicate that dissolved nitrogen generally affects fish when saturation exceeds 110%–130%, with the threshold level depending on water depth (Rucker and Tuttle 1948; Harvey and Cooper 1962; Fickeison et al. 1973; Blahm 1974; Meekin and Allen 1974; Meekin and Turner 1974; Rucker and Kangas 1974; Blahm et al. 1975; Dawley et al. 1975; Weitkamp 1975; Bentley and Dawley 1976; Bouck et al. 1976; Nebeker and Brett 1976). Gas bubble disease (GBD) and mortality are the primary detrimental effects associated with dissolved nitrogen concentrations above threshold levels (Parametrix 1975).The detrimental effects of nitrogen supersaturation vary according to the length of exposure (Blahm et al. 1975). Chinook salmon, coho salmon, steelhead trout, rainbow trout, whitefish, and largemouth bass were exposed to nitrogen levels of 130% for 8 of every 24 hours. Mortality did not exceed 50% if fish were placed in nitrogen-saturated water (i.e., 100%) of the remaining 16 hours. However, when fish were continuously exposed to supersaturated levels of nitrogen (130%), mortality rates exceeded 50% during the first day. Various species of juvenile salmonids may compensate for total nitrogen saturation levels up to 125% by remaining in deeper water (Parametrix 1975). Hydrostatic pressure increases with depth, so in deeper water nitrogen remains in solution in the blood of fish, inhibiting GBD.

Nutrients

Nutrient levels should remain within the natural range for the area and season as well as sustain the normal level of primary production. Various inorganic constituents of surface water are nutrients required for biological processes. Nitrogen and phosphorus are clearly the most important nutrients affecting productivity of aquatic systems. Natural sources of nitrogen and phosphorous in natural ecosystems are discussed at length in Section 3.8.1. Inputs to surface- and groundwaters can be affected by vegetation changes associated with land-use activities as well as through direct enrichment from sewage effluents, run-off from agricultural lands, and industrial water.

Nitrogen generally occurs in natural waters as nitrite or nitrate anions, as cationic forms like ammonium, and as intermediate oxidation states like those that occur in biological materials (e.g., decomposing organic solutes). In surface waters or in groundwaters that are impacted through human use, cyanide from industrial sources, mines, and various other complex effluents (including agricultural runoff) may also be sources of nitrogen. Nitrite-nitrogen is short-lived in natural environments and, consequently, rarely exists in concentrations toxic to salmonids. Acute toxicity values for salmonids range from 100 to 900 ppb as NO_2-N (48-hour or 96-hour LC_{50}); chronic effects are poorly understood, in part, because nitrite does not persist in surface waters under natural conditions.

Nitrate is formed by the complete oxidation of ammonia through the nitrification process and can be found in relatively high concentrations in surface waters. Unlike ammonia and nitrite, nitrate does not form un-ionized species in aqueous solutions and is considered essentially nontoxic for aquatic vertebrates and invertebrates (e.g., acute LC„ greater than 1300 ppm for salmonids). However, much lower concentrations of nitrate may lead to adverse effects associated with eutrophication and the development of oxygen-depleted waters (Leidy 1980; Rand and Petrocelli 1985).

Ammonia frequently acts as a toxicant in surface waters subject to high inputs of nitrogen, especially through anthropogenic activities (e.g., agricultural runoff, sewage effluents). For salmonids, ammonia is acutely toxic at concentrations as low as 80 ppb, but the initiation of ammonia toxicosis is highly variable, primarily as a function of pH. Physiological responses to ammonia exposure are frequently exacerbated by low dissolved oxygen concentrations; for salmonids, acute toxicity is increased two-fold when dissolved oxygen is decreased from 80% to

30% saturation. In the laboratory, chronic effects of ammonia have been documented as low as 2 ppb, but little work has been completed to identify the effects of long-term exposures under field settings (Rand and Petrocelli 1985; Reader and Dempsey 1989).

In contrast to nitrogen, phosphorus does not leach as readily from soil. In natural waters, phosphorus occurs in very low concentrations, most often in tenths of a milligram per liter (or less). Orthophosphate and its intermediates most frequently occur in surface waters and are routinely measured as "total phosphorus" in water-quality monitoring activities. Phosphorus most frequently occurs in surface waters as phosphates, which are generally considered nontoxic to aquatic vertebrates and invertebrates (Stumm and Morgan 1981; EPA 1986).

Nitrogen and phosphorus are the principal causes of nutrient enrichment of surface waters. Aquatic macrophytes (rooted-submerged and floating vascular plants) and algae are dependent to varying degrees on dissolved nitrogen and phosphorus for their nutrient supply. Growth of benthic algae and phytoplankton is particularly sensitive to the ratio of nitrogen to phosphorus. Enrichment leads to high production rates of biomass (e.g., algal blooms) that are undesirable for other aquatic biota, especially when respiration and decomposition create high biochemical oxygen demand and oxygen depletion. While the enhanced growth rates of aquatic vegetation can reach maximal conditions under nitrogen and phosphorus enrichment, phosphorus frequently acts as the limiting factor in aquatic habitats and will tend to control production rates (Leidy 1980; Stumm and Morgan 1981; Hem 1992).

Biocides

Agricultural chemicals are potentially widespread in the environment, and surface waters and groundwaters may be affected by chemical use that accompanies changes in land-use practices. Various classes of chemicals are currently used in the agricultural industry, including herbicides, insecticides, fungicides, nematicides, defoliants, rodenticides, and growth regulators. These are primarily organic chemicals, but inorganic chemicals, such as mineral salts and nutrients, may also be used as fertilizers and may directly effect receiving waters. Similarly, complex chemical mixtures from industries, municipalities, and landfills may impact water resources through runoff or infiltration to groundwater (Leidy 1980; Rand and Petrocelli 1985).

Agricultural chemicals are regulated to decrease the likelihood of their release into surface waters and groundwaters, and water-quality criteria have been established for many of these chemicals (Table 5-1). There are several properties of organic chemicals that influence their fate and effects in the environment. For surface waters and groundwaters, a chemical's adsorptivity, stability, solubility, and toxicity will determine the extent to which that chemical will migrate and adversely effect a water resource. Among the thousands of agricultural chemicals available for users (industries, small businesses, farmers, orchardists, and home gardeners), these properties will vary significantly. Depending upon the chemical's physicochemical properties, the potential contamination of water resources may be complex. For example, a chemical's water solubility will influence whether it occurs in solution or adsorbed to sediments or colloids held in suspension (Dickson et al. 1987; Rand and Petrocelli 1985). Synthetic organics, even at subacute levels, may alter neurological, endocrine, and behavioral functions in fish (e.g., Folmar 1993; Choudhary et al. 1993; Singh et al. 1994). In addition to being toxic to fish and invertebrates that fish eat, organic chemicals may indirectly affect nontarget species through habitat alteration (e.g., changes in plant community structure as a result of targeting weedy species), and such changes may occur even under the best management practices (Leidy 1980).

Heavy Metals

Metal concentrations in surface water vary regionally, reflecting the geochemical composition of the underlying parent material and the soils characteristic of the watershed. Most frequently, metals occur in trace quantities as a result of soil leaching and geochemical processes that occur in the underlying bedrock. The concentration of metals in surface waters may be increased by anthropogenic activities such as mining and related industrial practices, such as electroplating and metals refining (Leidy 1980; Stumm and Morgan 1981; Rand and Petrocelli 1985).

Although some metals are necessary trace nutrients, many metals are toxic to fish at very low concentrations. Other water-quality conditions influence the bioavailability of the metals. For example, metals that are nutritional requirements must be absorbed by the organism. Metals may occur in solution and may be available for uptake directly from the water, or they may be adsorbed to colloidal particles in the water column. The extent to which metals are adsorbed and then intentionally or coincidentally ingested may influence the onset of metal toxicosis in aquatic biota, especially when the interaction between the metals in solution and metals adsorbed to colloids of various forms (e.g., relatively simple organic ligands versus complex organic structures like the humic acids) is influenced by other water-quality conditions such as pH. Table 5-2 lists water-quality criteria for selected metals and metalloids that are frequently considered toxicants of

Table 5-1. Water-quality criteria for selected herbicides, pesticides, and fungicides in freshwaters. From EPA (1986).

Chemical	Maximum acceptable levels (μg/L)	
	Acute (instantaneous)	Chronic (24-hour average)
Aldrin	3.0	
Chlordane	2.4	0.0043
Chlorophenoxy herbicides		
2,4-D	100.0	
2,4,5-TP	10.0	
Chloropyrifos	0.083*	
DDT and metabolites		
DDT	1.10†	0.001†
TDE	0.60†	
DDE	1050.0†	
Dieldrin	2.5	0.0019
Endosullfan	0.22	0.056
Endrin	0.18	0.023
Guthion	0.01	
Heptachlor	0.52	0.0038
Lindane	2.0	0.080
Malathion	0.10	
Methoxychlor	0.012	0.030
Mirex	0.001	
Parathion	0.04	
Pentachlorophenol	‡	
Toxaphene	1.60	0.013

* I-hour average, not more than 1 time per 3 years.
† Human-health based criteria.
‡ Criteria based on pH: see current EPA criteria.

Table 5-2. Water-quality criteria for metals and metalloids found in surface waters. Criteria for most metals are a function of water hardness. Also shown are other factors affecting toxicity and anthropogenic sources. From EPA (1986).

Metal	Anthropogenic source	Maximum acceptable levels (μg/L)		Other factors influencing toxicity
		4-day average*	1-hour average*	
Arsenic	Agrichemicals	190	360	
Boron	Agrichemicals	750†		
Cadmium	Mining/industrial	$e^{(1.128[\ln(\text{hardness})]-3.49)}$	$e^{(1.128[\ln(\text{hardness})]-3.828)}$	pH
Chromium VI	Industrial	11	16	pH
Chromium III	Industrial	$e^{(0.819[\ln(\text{hardness})]+1.561)}$	$e^{(0.819[\ln(\text{hardness})]+3.688)}$	pH
Copper	Mining/industrial	$e^{(0.8545[\ln(\text{hardness})]-1.465)}$	$e^{(0.9422[\ln(\text{hardness})]-1.464)}$	pH, valence, temperature, other metals
Iron	Mining/industrial		1,000#	
Lead	Mining/industrial	$e^{(1.273[\ln(\text{hardness})]-4.705)}$	$e^{(1.273[\ln(\text{hardness})]-1.460)}$	Turbidity, pH
Manganese	Industrial	50‡		
Mercury	Industrial	0.144§		DOC, microbial activity
Nickel	Mining/industrial	$e^{(0.76[\ln(\text{hardness})]+1.06)}$¶	$e^{(0.76[\ln(\text{hardness})]+4.02)}$#	
Selenium	Irrigated agriculture	35¶	260#	
Silver	Mining		$e^{(1.72[\ln(\text{hardness})]-6.52)}$#	
Zinc	Mining/industrial		$e^{(0.83[\ln(\text{hardness})]+1.95)}$#	pH, temperature, valence

* Values not to be exceeded more than 1 time per 3 years.
† Criteria is for long-term irrigation of crops. No freshwater standards given
‡ Domestic water supplies.
§ Human health criteria.
¶ 24 hour average.
\# Instantaneous.

concern, especially in surface waters impacted by human use. The physical features of the surrounding habitat (e.g., land-use in riparian areas) may influence a chemical's toxicity, and seasonal variations in bioavailability of contaminants (e.g., changing redox potentials of sediments and availability of metals) must also be considered (Stumm and Morgan 1981; Dickson et al. 1987; Adriano 1992).

pH

Acidic surface waters may occur naturally as a result of dissolution of parent materials in bedrock and overlying soils, biological decomposition (especially processes yielding organic acids such as fulvic and humic acids), or through geothermal activity or catastrophic events related to volcanic activity. More frequently however, surface-water acidity results from anthropogenic activities related to land use (e.g., mining) or resource use (e.g., combustion of fossil fuels) with the subsequent deposition of materials capable of generating—directly or indirectly—and releasing hydrogen ions to the environment (Leidy 1980; Rand and Petrocelli 1985). The influence of hydrogen ions on aquatic organisms is influenced by watershed characteristics, including the buffering capacity of soils as well as by concentrations of dissolved materials in surface waters (Rand and Petrocelli 1985; Brown and Sadler 1989).

In general, fish may be adversely affected by surface water with pH 5.6 or less; however, the threshold for adverse effects is species-specific and water-quality dependent (e.g., buffering capacity). Hence, no single pH value can be regarded as a threshold for anticipating population-level responses to acidic surface waters. Respiratory problems are frequently observed in experimental fish exposed to low pH. Mucous clogging, increased ventilation, coughing and hypoxia are commonly recorded in acid-exposed fish. Aluminum and other metals exacerbate the physiological response to increased hydrogen ion. Low pH alters the specific form of metals in soils, increasing both their mobility and their bioavailability to aquatic organism. In addition, low pH acts synergistically with heavy metals in surface waters to yield adverse biological effects (Stumm and Morgan 1981; Rand and Petrocelli 1985; Brown and Sadler 1989). High pH values may also adversely affect salmonids. Elevated pH can arise when reductions in canopy cover in riparian zones stimulates production of algae. **As** algae photosynthesize during the day, they take up carbon dioxide, which results in a reduction in free hydrogen ions (i.e., increasing pH).

5.1.3 Habitat Structure

The physical structure of streams, rivers, and estuaries plays a significant role in determining the suitability of aquatic habitats to salmonids as well as other organisms upon which salmonids depend for food. These structural elements are created through interactions between natural geomorphic features, the power of flowing water, sediments that are delivered to the stream channel, and riparian vegetation, which provides bank stability and inputs of large woody debris. Structural attributes of streams vary naturally among regions and along the longitudinal dimension of streams in response to differences in topography, geology, geomorphic features, hydrologic regime, sediment load, and riparian vegetation (see Sections 3.5 and 3.9.5). These spatial differences and gradients give rise to the variety of macro- and microhabitat attributes that are used by salmonids at various stages of their life histories. Macrohabitat features include pools, glides, and riffles. The relative frequency of these habitat types changes with size of the stream, the degree of channel constriction, and the presence of large woody debris. Microhabitat attributes include characteristics such as substrate type, cover, depth, hydraulic complexity, and current velocity.

Because of the great diversity in the physical attributes of western streams and in the requirements of various salmonids, and because few undisturbed watersheds remain to serve as reference points, it is difficult to quantify natural ranges of physical habitat features in streams, rivers, and estuaries. For example, historically, mid-order streams west of the Cascade crest had 16–38 pools per km (25–60 per mile) (FEMAT 1993). Pool frequencies in 10 human-influenced tributaries of the upper Grande Ronde River ranged from 3.8–26.2 per km (6–42 per mile) in 1941 and 1.4–7.4 per km (2–12 per mile) in 1990 (McIntosh et al. 1994b). In the Yakima Basin, an unmanaged watershed (Rattlesnake Creek) averaged 1.6 pools per km (2.5 per mile) in 1935–1936 and 3.9 pools per km (6 per mile) in 1987–92 (McIntosh et al. 1994b); similar pool frequencies were also reported for the Chewack River in the Methow River Basin. In low gradient streams on the Olympic Peninsula, Washington, pools constituted 81.1% of the stream surface area (160 pools $> 10\ m^2$ per mile) (Grette 1985). In low-gradient stream reaches in southeast Alaska, pools accounted for 39%–67% of the surface area depending upon bank full width (Murphy et al. 1984 discussed in Peterson et al. 1992). This high degree of variation illustrates the importance of local geomorphic features, stream size, and riparian influence on stream habitat characteristics.

Despite inherent differences in streams, it is clear that habitat complexity is an important feature of aquatic systems. In streams of the Pacific Northwest, large woody debris creates both macro- and microhabitat complexity that is essential to salmonids and other aquatic organisms. Large wood creates habitat heterogeneity by forming pools, back eddies, and side channels and by creating channel sinuosity and hydraulic complexity. Large wood also functions to retain coarse sediments (e.g., spawning gravels) and organic matter in addition to providing substrate for numerous aquatic invertebrates. McIntosh et al. (1994b) reported that changes in substrate composition towards smaller fractions coincided with reduced frequency of large woody debris in streams of the upper Grande Ronde River. Consequently, large woody debris plays a significant part in controlling other structural elements of streams.

Large woody debris provides an important component to estuarine habitats of coastal rivers (Maser et al. 1988), which are important rearing areas for juvenile anadromous salmonids (Table 4-2). Woody debris increases habitat complexity in areas where the bottom consists mainly of fine sediments. Numerous invertebrates rapidly process the wood, liberating nutrients for some organisms, while others use the wood as refugia. In salt marshes, large woody debris traps sediments to increase the extent of the marsh. As exceptionally high tides displace the logs, depressions left in the sediments increase habitat diversity important to juvenile fishes. In areas that are predominantly mud bottomed, large wood further serves as a repository for herring spawn. The functional roles of large woody debris in streams, and how these change from headwater reaches to estuaries, are reviewed in greater detail in Section 3.9.5. The functions of large woody debris relative to specific life stages of salmonids are discussed in Section 5.2.

Other important components of habitat structure at the microscale include large boulders, coarse substrate, undercut banks, and overhanging vegetation. These habitat elements offer salmonids concealment from predators and shelter from fast currents. At the macrolevel, streams and rivers with high channel sinuosity, multiple channels and sloughs, beaver impoundments, or backwaters typically provide high-quality habitat for salmonids. Such areas serve as refugia during high flows. Salmonids in estuaries benefit from similar structural features (substrate complexity, overhanging vegetation, depth heterogeneity) as well as abundant macrophytes.

5.1.4 Flow Regime

Flow regimes in streams and rivers determine the amount of water available to salmonids and other aquatic organisms, the types of micro- and macrohabitats that are available to salmonids (see Section 5.2), and the seasonal patterns of disturbance to aquatic communities. High-flow events redistribute sediments in streams, flushing fine sediments from spawning gravels and allowing recruitment of gravels to downstream reaches. In addition, extreme flow events are essential in the development and maintenance of healthy floodplain systems through deposition of sediments, recharge of groundwater aquifers, dispersal of vegetation propagules, recruiting large woody debris into streams, and transporting wood downstream. In alluviated reaches, high flows may create new side channels and flood off-channel areas that are important rearing habitats for salmonids. Low flow may also be important for the establishment of riparian vegetation on gravel bars and along stream banks (Section 3.6). Thus, although over shorter time scales high- or low-flow events may temporarily reduce salmonid numbers, dynamic flows are needed to perform essential functions important in the long-term persistence of salmonid populations.

The specific flow requirements of salmonids vary with species, life history stage, and time of year (see Section 5.2). Local salmonid populations have evolved behavioral and physical characteristics that allow them to survive the flow regimes encountered during each phase of their development. Protection of salmonid habitats requires streamflows to fluctuate within the natural range of flows for the given location and season.

5.1.5 Biotic Interactions

Protecting and restoring biological integrity in surface waters also depends on maintaining natural biological interactions among species. These interactions may be affected directly by the introduction of non-native species and stocks (see Sections 4.3 and 6.12) and overexploitation (Section 6.11) or indirectly through modification of physical and chemical characteristics of streams, lakes, and estuaries (reviewed in Chapter 4). Human-induced impacts on biological interactions include changes in primary and secondary production, disruption in timing of life history events or seasonal rhythms, increased frequency of disease or parasitism, and changes in the outcome of predator-prey and competitive interactions. Together these perturbations lead to changes in food webs and trophic structure of aquatic systems.

5.2 Habitat Requirements by Life Stage

Salmonids use a variety of habitats during their life histories. Anadromous species in particular have complex life histories that involve periodic shifts in habitat. Depending on the species or stock, freshwater streams, lakes, or intertidal sloughs may be used for reproduction; streams, lakes, estuaries, or oceans may be used for juvenile rearing (Table 4-2). For all anadromous species, habitats between spawning streams and the ocean are required for upstream and downstream migrations. Differences in spatial and temporal use of specific habitats exist for each species, yet the diversity among species and by life stage indicates that most freshwater habitats are utilized year round (Table 4-3). Juvenile-to-adulthood rearing generally occurs in the ocean, but there is considerable variation (Table 4-2), even within each species. To persist, each species or stock must be able to survive within the entire range of habitats encountered during its life; degradation or alteration of habitat required at any life stage can limit production. Much of the available information on salmonid habitat requirements has been summarized in reviews by Bell (1986), Everest et al. (1985), and Bjornn and Reiser (1991), which are the primary sources of information for this section unless otherwise noted.

Most of the quantitative descriptions of requirements for salmonid habitats presented in this section consist of either microhabitat observations of salmonids in nature or results from laboratory studies that measure the performance of salmonids (often hatchery fish) under controlled conditions. Microhabitat measurements are frequently made during a single season (usually summer, when sampling is easiest), and the resulting data are often reported in the literature without accompanying data on habitat availability. Habitat utilization constitutes a "preference" only when the particular range of depths, velocities, substrates, or cover types is used at a frequency greater than its general availability in the environment. The range of microhabitats used in a particular stream depends on availability. Consequently, variation in microhabitat utilization among streams (or segments of the same stream) can be substantial, and habitat preference information derived from one location should not be applied to other areas without careful consideration of similarities and differences between sites. In addition, microhabitat measurements at holding positions of salmon and trout do not always encompass the range of velocities needed for feeding, which are commonly higher. For all of these reasons, care must be taken when interpreting microhabitat data published in the literature. Similarly, optimal conditions for development, growth, and survival as determined in the laboratory do not always correspond to the most favorable conditions in natural environments (see Section 5.1).

5.2.1 Adult Migrations

The migrations of anadromous salmonids from river mouths to their natal streams vary in length from a few hundred meters (e.g., chum salmon spawning in the intertidal zone) to well over a thousand kilometers. Even resident fish may make substantial migrations between lakes and streams or between sections of a river network (Everest et al. 1985). During upstream migrations, anadromous salmonids need holding or resting sites and suitable flow and water quality. Resident species may feed during their migrations and thus may have more diverse habitat needs.

Physical Structure

Upstream migration of many salmonid species typically involves rapid movements through shallow areas, followed by periods of rest in deeper pools. Some races, such as spring chinook and summer steelhead, may arrive at spawning sites several months before spawning or hold in mainstem rivers for several weeks or months prior to moving into their natal streams to spawn (Bjornn and Reiser 1991). Large woody debris, boulders, and other structures provide hydraulic complexity (e.g., eddies or localized areas of slow water) and pool habitats that serve as resting stations for fish as they migrate upstream to spawn. Resident species use structure to pause out of the main current while waiting for prey to drift by in adjacent, faster waters. Large woody debris and other structures may also facilitate temperature stratification and the development of thermal refugia by isolating pockets of cold water and preventing mixing (see discussion of temperature below). In shallower reaches, riparian vegetation and large wood provide cover from terrestrial predators. At redd sites, adequate areas of stable, appropriately sized gravel containing minimal fine sediments are required for successful spawning (see Section 5.2.2).

Flows and Depth

Streamflow during the spawning migration must be sufficient to allow passage over physical barriers including falls, cascades, and debris jams; as a result, the migrations of many stocks occur coincident with high flows. Coho salmon frequently wait near stream mouths until a freshet occurs before moving upstream (Sandercock 1991), as may pink salmon (Heard 1991). Holtby et al. (1984) observed continuous entry of coho salmon into Carnation Creek during years of high flow but pulsed entry when freshets were infrequent. Spring and summer

chinook stocks migrate during periods of high flows that allow them to reach spawning tributaries in headwater reaches, while fall-run stocks, which typically spawn in lower reaches, may enter streams during periods of relatively low flow (Healey 1991).

Minimum depths that will allow passage of salmonids are approximately 12 cm for trout, 18 cm for the smaller anadromous species (i.e. pink, chum, steelhead, sockeye, and coho salmon), and 24 cm for large chinook salmon (Bjornn and Reiser 1991); however, substantially greater depths may be needed to negotiate larger barriers. Reiser and Peacock (1985) report that maximum leaping ability varies from 0.8 m for brown trout to 3.4 m for steelhead. Pool depths must exceed barrier height by approximately 25% to allow fish to reach the swimming velocities necessary to leap to these heights (Stuart 1962). The ability to pass a barrier is also influenced by pool configuration. Water plunging over a steep fall forms a standing wave that may allow salmonids to attain maximum heights (Bjornn and Reiser 1991). Less severe inclines (e.g., cascades) may be more difficult to pass if pool depths are inadequate and velocities are high.

Water Quality

Temperature. Most adult salmonids typically migrate at temperatures less than 14"C; however, spring and summer chinook salmon migrate during periods when temperatures are substantially warmer (Table 5-3). Excessively high or low temperatures may result in delays in migration (Major and Mighell 1966; Hallock et al. 1970; Monan et al. 1975). Adult steelhead that move from the ocean into river systems in the summer and fall may overwinter in larger rivers, delaying entry into smaller spawning tributaries until they are free of ice in the spring. Similarly, spring-spawning resident salmonids, including cutthroat and rainbow trout, may hold at the mouths of spawning streams until temperatures warm up to the preferred temperature range (Bjornn and Reiser 1991). In addition to delaying migration, excessively high temperatures during migration may cause outbreaks of disease (see Section 4.3.4).

Coldwater refugia may also be important to adult salmon as they migrate upstream. Adult summer-run steelhead in the Middle Fork Eel River of California were observed in thermally stratified pools, but were absent or infrequent in non-stratified pools of similar depth (Nielsen et al. 1994). Coldwater pockets in stratified pools ranged from 4.1 to 8.2°C cooler than ambient stream temperatures. Spring chinook salmon have also been observed to hold in coldwater pools for several months prior to spawning in the Yakima River of eastern Washington, moving as much as 60 km from holding pools to spawning sites (NRC 1992). The authors suggest that this behavioral thermoregulation lowers metabolic rates and thereby conserves energy for gamete production, mate selection, redd construction, spawning, and redd guarding.

Streamflow, channel morphology, and the presence of large woody debris may play significant roles in mediating the formation and persistence of coldwater refugia (Bilby 1984; Nielsen et al. 1994). In some streams and rivers, gravel bars or other structures isolate incoming tributaries or seep areas from mainstem waters, thereby inhibiting the mixing of waters and helping to maintain thermal gradients (Nielsen et al. 1994). In larger systems, thermally stratified pools need not be associated with coldwater inputs provided that deep scour pools exist and flows are sufficiently low to prevent turbulent mixing. Consequently, in larger systems management practices that reduce large woody debris, destabilize stream channels, increase turbulence or modify stream flows may eliminate coldwater refugia.

Dissolved Oxygen. The high energy expenditures of sustained upstream swimming by salmonids requires adequate concentrations of dissolved oxygen (DO). Davis et al. (1963) found adult and juvenile swimming performance impaired when DO dropped below 100% saturation levels for water temperatures between 10–20°C. DO concentrations below 6.5–7.0 mg/L greatly impaired performance at all temperatures studied. Migrating adults exhibited an avoidance response to DO levels below 4.5 mg/L (Hallock et al. 1970). Migration resumed when DO levels increased to 5 mg/L.

Turbidity. High concentrations of suspended sediment may delay or divert spawning runs and in some instances can cause avoidance by spawning salmon (Smith 1939; Servizi et al. 1969; Mortensen et al. 1976). Salmonids were found to hold in a stream where the suspended sediment load reached 4,000 mg/L (Bell 1986). Though high sediment loads may delay migration, homing ability does not seem to be adversely affected (Murphy 1995). Cowlitz River chinook salmon returned to the hatchery seemingly unaffected by the sediments derived from the eruption of Mount St. Helens, Washington, although in the highly impacted Toutle River tributary of the Cowlitz, coho salmon did stray to nearby streams for the first two years following the eruption (Quinn and Fresh 1984).

Table 5-3. Tolerable and preferred temperature ranges (°C) for adult migration, spawning, and incubation of embryos for native salmonids in the Pacific Northwest. Modified after Bjornn and Reiser (1991).

Species	Life Stage		
	Spawning Migration (min – max)	Spawning (preferred range)	Incubation (preferred range)
ANADROMOUS			
Pink salmon	7.2 – 15.6*	7.2 – 12.8*	4.4 – 13.3*
Chum salmon	8.3 – 15.6*	7.2 – 12.8*	4.4 – 13.3*
Coho salmon	7.2 – 15.6*	4.4 – 9.4*	4.4 – 13.3*
Sockeye salmon	7.2 – 15.6*	10.6 – 12.2*	4.4 – 13.3*
Spring chinook	3.3 – 13.3''	5.6 – 13.9*	5.0 – 14.4*
Summer chinook	13.9 – 20.0*	5.6 – 13.9*	5.0 – 14.4*
Fall chinook	10.6 – 19.4*	5.6 – 13.9*	5.0 – 14.4*
Steelhead trout		3.9 – 9.4*	
Cutthroat trout		6.1 – 17.2*	
RESIDENT			
Kokanee		5.0 – 12.8*	
Mountain whitefish		0.0 – 5.6†	
Cutthroat trout	5.0 – 10.0	4.4 – 12.8† 5.5 – 15.5‡	
Rainbow trout		2.2 – 20.0* 4.4 – 12.8†	
Dolly Varden		7.8†	
Bull trout		< 9.0§ 4.5¶	2.0 – 6.0§

* Bell 1986.
† Everest et al. 1985.
‡ Varley & Gresswell 1988.
§ Pratt 1992.
¶ Ratliff 1992.

5.2.2 Spawning and Incubation

Although spawning and incubation occur in the same habitat, adults and embryos have slightly different habitat needs. Adults select sites based on substrate composition, cover, and water quality and quantity. Embryo survival in and fry emergence from an intragravel environment depends upon physical, hydraulic, and chemical variables including substrate size, channel gradient and configuration, water depth and velocity, DO, water temperature, biochemical oxygen demand in the gravel, and permeability and porosity of the gravel in the redd (Bjornn and Reiser 1991).

Physical Structure

All salmonids require sufficient gravels within a specific size range and a minimum of fine sediments for successful spawning. Usable gravel size generally is proportional to adult size—larger individuals spawn in larger substrate (Marcus et al. 1990). Bjornn and Reiser (1991) reviewed the available literature and found that anadromous salmon typically use gravels in the 1.3–10.2-cm size range, whereas steelhead and resident trouts may use smaller substrates (0.6–10.2 cm). The depth that salmonids deposit eggs within the substrate is also a function of size (Everest et al. 1985) and may be critical to incubation success. Nawa and Frissell (1993) found that gravel beds can be both scoured and filled during the same flood event potentially leaving little net change in bed surface elevation. Eggs deposited within the zone of scour and fill are likely to wash downstream. Bedload and bank stability arising from LWD and intact upslope, floodplain, or riparian zones minimize this risk. Large woody debris diversifies flows, reducing stream energy directed towards some portions of the stream (Naiman et al. 1992). This creates pockets of relatively stable gravels better protected from the scouring effects of high flows.

Flow and Depth

The number of spawning salmon and trout that can be accommodated in a given stream depends on the availability of suitable habitats for redd construction, egg deposition, and incubation (Bjornn and Reiser 1991). Two characteristics of spawning habitats directly tied to streamflow are water depth and current velocity. Salmonids typically deposit eggs within a range of depths and velocities that minimize the risk of desiccation as water level recedes and that ensure the exchange of water between surface and substrate interstices is adequate to maintain high oxygen levels and remove metabolic wastes from the redd. In general, the amount of habitat suitable for spawning increases with increasing streamflow; however, excessively high flows can cause scouring of the substrate, resulting in mortality to developing embryos and alevins (Hooper 1973).

Bjornn and Reiser (1991) recently reviewed studies quantifying specific water depths and velocities at sites used by salmonids for spawning in rivers and streams. In Table **5-4,** results from their review have been supplemented with data from four other reviews (Healey 1991; Heard 1991; Salo 1991; and Sandercock 1991) on spawning sites for anadromous salmonids. Usually, depth and velocity of water at spawning sites is related to the size of spawners: larger species spawn at greater depths and faster water velocities than smaller species. There is also substantial variation among rivers, probably reflecting differences in habitat availability. Most species typically spawn at depths greater than 15 cm, with the exception of kokanee salmon and smaller trout (Table 5-4), which spawn in shallower waters. Location of redd sites based on water depths and velocities may also vary depending on spawner density. For example, pink salmon tend to spawn in shallower waters when conditions are crowded or streamflow is low (Heard 1991). Several species of salmonids may seek out areas of upwelling for spawning; these include sockeye salmon, chum salmon, coho salmon, and bull trout (Burgner 1991; Salo 1991; Sandercock 1991; Pratt 1992). Upwelling increases circulation of water through redds, which helps to eliminate wastes and prevents sediments from filling in spawning gravel interstices. Thus infiltration that recharges groundwater, which eventually discharges in subsurface springs and seeps, must be maintained.

Water Quality

Temperature. Salmonids have been observed to spawn at temperatures ranging from 1–20°C (Bjornn and Reiser 1991), but most spawning occurs at temperatures between 4 and 14°C (Table 5-3). Resident trouts, including rainbow and cutthroat trout, may spawn at temperatures up to 20.0°C and 17.2°C, respectively, while coho salmon, steelhead trout, Dolly Varden, bull trout, and mountain whitefish tend to prefer lower temperatures. The wide range of spawning temperatures utilized by most salmonid species strongly suggests that adaptation has allowed salmonids to persist in a variety of thermal environments and that attempting to identify species-specific preferenda may fail to account for ecological requirements of individual stocks.

Among the salmonids, the preferred incubation temperatures have been best documented for the anadromous species. Bell (1986) suggested preferred temperature ranges of 4.4–13.3°C for pink salmon, chum salmon, coho salmon, and sockeye salmon, and 5.0–14.4°C for chinook salmon (Table 5-3). More

Table 5-4. Water depths and velocities used by anadromous and resident salmonids for spawning.

Species	Depth (cm)	Velocity (cm/s)	Source
Chinook salmon (race not specified)	15 – 43*	37 – 69*	Bovee (1978)
	52 – 128†	55 – 1137	Graybill et al. (1979)
	30 – 460		Chapman (1943)
Spring chinook salmon	≥ 24	30 – 91	Thompson (1972)
	18 – 38*	24 – 61*	Bovee (1978)
	5 – 122		Burner (1951)
	13 – 720	30 – 150	Vronskiy (1972)
	45 – 52	52 – 68	Collings et al. (1972)
		22 – 64	Smith (1973)
	30 – 107	30 – 53	Chambers et al. (1955)
		15 – 100	Neilson and Banford (1983)
Summer chinook salmon	≥ 30	32 – 109	Reiser and White (1981)
	5 – 700	10 – 189	Healey (1991)
Fall chinook salmon	10 – 120	25 – 115	Bovee (1978)
	≥ 24	30 – 91	Thompson (1972)
	122 – 198	84 – 114	Chambers et al. (1955)
	28 – 41	30 – 76	Briggs (1953)
	30 – 45	30 – 68	Collings et al. (1972)
		19 – 81	Smith (1973)
	to 700	37 – 189	Chapman et al. (1986)
Chum salmon	≥ 18	46 – 101	Smith (1973)
	13 – 50†	21 – 84†	Johnson et al. (1971)
	20 – 110	10 – 20	Sano and Nagasawa (1958)
	30 – 100	10 – 100	Soin (1954)
Coho salmon	≥ 18	30 – 91	Thompson (1972)
	4 – 33	30 – 55	Gribanov (1948)
	12 – 35*	25 – 61*	Bovee (1978)
	20 – 25	25 – 70	Li et al. (1979)
	10 – 20	30 – 75	Briggs (1953)
Pink salmon	≥ 15	21 – 101	Collings (1974)
	10 – 150	30 – 140	Heard (1991)
Sockeye salmon	≥ 15	21 – 101‡	Bjornn and Reiser (1991)
	15 – 300		Burgner (1991)
	17 – 49††	34 – 58*	Bovee (1978)
	15 – 55‡‡	28 – 79‡‡	Stober and Graybill (1974)
	30 – 46	53 – 55	Clay (1961)
Kokanee salmon	≥ 6	15 – 73	Smith (1973)
	6 – 23*	11 – 41*	Bovee (1978)
Steelhead trout (race not specified)	≥ 24	40 – 91	Smith (1973)
	18†	30 – 91†	Stober and Graybill (1974)
	12 – 70	37 – 109	Hunter (1973)
	27 – 88t	46 – 91†	Graybill et al. (1979)
Winter Steelhead trout	24 – 55*	43 – 87*	Bovee (1978)
Rainbow trout	≥ 18	48 – 91	Smith (1973)
	15 – 43	27 – 79	Chambers et al. (1955)
	21 – 30	30	Li et al. (1979)
Cutthroat trout	≥ 6	11 – 72	Hunter (1973)
	17 – 30	15 – 46	Chambers et al. (1955)
Mountain whitefish	≥ 23	30 – 66*	Bovee (1978)
	610 – 1220	≥ 15	Li et al. (1979)

* Values indicate 50% probability range
† Values indicate 80% probability range.
‡ Estimated by Bjornn and Reiser (1991) based on criteria for other species.

recent laboratory studies have demonstrated that coho and sockeye salmon embryos tend to be less sensitive to cold temperatures and more sensitive to warm temperatures than pink, chum, or chinook salmon (Murray and McPhail 1988; Beacham and Murray 1990). Coho and sockeye salmon embryos incubated at 1.0°C had survival rates higher than 50%; chum and chinook salmon embryos exhibited 50% mortality at temperatures below 2.5 and 3.0°C, respectively; and even and odd-year pink salmon exhibited 50% mortality at 3.5 and 4.5°C, respectively (Beacham and Murray 1990). Conversely, 50% mortality occurred at temperatures above 13.5°C for coho salmon embryos, compared with 15–15.5°C for pink and sockeye salmon, and 16°C for chum and chinook salmon. The alevin stage is generally less temperature sensitive than the embryonic stages, with lower low-temperature thresholds, and higher high-temperature thresholds (Beacham and Murray 1990). Salmonid embryos and alevins can tolerate short periods during which temperatures are below or above incipient lethal levels (Bjornn and Reiser 1991).

Seymour (1956) carried out comprehensive studies on temperature effects on the development of chinook salmon from the egg to fingerling stage. Environmental temperature was correlated with the number of vertebrae, egg mortality, the number of abnormal fry, and the duration of the hatching period. For eggs reared at temperatures between 4.4 and 14.4°C, no differences were observed, but defects and mortality increased at both higher and lower temperatures. Combs (1965) identified lower (4.4–5.8"C) and upper (12.7–14.2"C) temperature thresholds for normal development of sockeye salmon eggs.

Dissolved Oxygen. Embryos and alevins need high levels of oxygen to survive (Shirazi and Seim 1981). Field studies have demonstrated positive correlations between DO and survival for steelhead trout (Coble 1961) and coho salmon (Phillips and Campbell 1961). Phillips and Campbell (1961) suggested that DO levels must average greater than 8.0 mg/L for embryos and alevins to survive well.

In addition to being directly lethal, low levels of dissolved oxygen can have sublethal affects on salmonids as well. The rate of embryological development, the time to hatching, and size of emerging fry are all affected by low levels of dissolved oxygen. Garside (1966) found that the rate of embryonic development was increasingly retarded by progressively lower levels of dissolved oxygen (DO), resulting in delayed hatching. Doudoroff and Warren (1965) reported that DO levels below saturation resulted in increases in time to hatching and completion of yolk-sac absorption, as well as decreases in the size of alevins. Silver et al. (1963) and Shumway et al. (1964) observed that steelhead trout, coho salmon, and chinook salmon reared in water with low or intermediate oxygen concentration were smaller in size and had a longer incubation period than those raised at high DO. Similarly, Brannon (1965) found a positive relationship between DO and the size of sockeye salmon alevins at time of hatching. Alderdice et al. (1958) found that very low oxygen levels at early egg incubation stages produced severe morphometric abnormalities in chum salmon in addition to delaying hatching. Low DO levels stimulated eggs in an advanced stage of development to hatch prematurely, causing mortality.

Bjornn and Reiser (1991), summarizing four different studies, concluded that critical dissolved oxygen levels needed to meet respiratory demands vary with state of development. Early embryological states (pre-eyed) require the lowest levels of oxygen, while embryos nearing hatching have the highest DO requirements.

Turbidify and Sedimentation. Salmonids require gravels that have low concentrations of fine sediments and organic material for successful spawning and incubation. Bedload or suspended organic and inorganic materials that settle out over spawning redds affect the intragravel environment of salmonid embryos in several ways. Inorganic sediments, as discussed above, may clog substrate interstices and thereby diminish intragravel flows. In addition fine sediments may act as a physical barrier to fry emergence (Cooper 1959, 1965; Wickett 1958; McNeil and Ahnell 1964; Koski 1972; Everest et al. 1987). Eggs deposited in small gravel or gravel with a high percentage of fine sediments have lower survival to emergence (Harrison 1923; Hobbs 1937; Shapovalov and Berrian 1940; Shaw and Maga 1943; Koski 1966). McHenry et al. (1994) found that excessive fines (> 13% of sediments < 0.85 mm) resulted in intragravel mortality for coho salmon and steelhead trout embryos because of oxygen stress. Organic materials that enter the substrate interstices use up oxygen as they decompose (Bjornn and Reiser 1991), further reducing DO concentrations. In addition, salmon and trout avoid areas with high percentages of sand, silt, and clay (Burner 1951; Stuart 1953).

5.2.3 Rearing Habitat: Juveniles and Adult Residents

The abundance of juveniles and resident adult salmonids is influenced by the quantity and quality of suitable habitat, food availability, and interactions with other species, including predators and competitors (Bjornn and Reiser 1991). **As** noted in Section 4.2, the types of rivers and streams used for

spawning and rearing differ among species and life stages. in addition, within the same system, sympatric populations of salmonids may segregate by channel unit types (e.g., pools, glides, riffles, cascades, off-channel areas) or by microhabitats within channel-unit types. The selection of specific microhabitats likely reflects a balancing among various factors, including the availability of food, the energetic costs of holding, risks of predation, and intra- and interspecific interactions. Bjornn and Reiser (1991) suggest that at any given time, certain environmental parameters may be better suited for some individuals, populations or species, while other parameters may not be as favorable yet must be kept in a suitable range for organism persistence. Consequently, there is no set of "optimal" habitat conditions for all species at all life stages.

Physical Structure

A variety of lentic and lotic habitats are potentially available to juvenile salmonids because of species-specific differences in ecological specialization. Comparison of habitat requirements among species is difficult: habitat selection is influenced by life stage, time of year, food availability, year-to-year variation in environmental conditions (e.g., flow, depth, temperature, food), and presence of other salmonids (Everest et al. 1985; Bjornn and Reiser 1991). Nevertheless, some species-specific affinities for habitats have been documented in the literature.

Shortly following emergence, the fry of many salmonids occupy shallow habitats along the margins of streams, moving into deeper and faster waters as they increase in size. Pink and chum salmon generally migrate to sea immediately after emergence. Sockeye salmon primarily use lakes as nursery areas, but will occasionally overwinter in sloughs, side channels, and spring areas (Burgner 1991). Within lakes, fry often use littoral areas for a month or more before moving offshore (Burgner 1991); riparian vegetation and woody debris may provide cover during this phase. Juvenile coho salmon tend to prefer pool habitats in summer and often move into side channels, sloughs, or beaver ponds for winter (Meehan and Bjornn 1991; Tschaplinski and Hartman 1983). Cutthroat trout also exhibit preference for pool habitats in summer, but the presence of other species such as coho salmon may cause trout to move into riffle habitats (Glova 1986). Steelhead trout typically prefer riffle habitats during summer (Everest et al. 1985) but may shift to pool habitats in winter or when coho salmon are not present (Bjornn and Reiser 1991). Juvenile chinook salmon are typically found in glide and riffle habitats with faster waters than typically used by coho salmon, though chinook do use pool habitats when available. Backwaters and side-channels that developed along unconstrained reaches in alluvial floodplains were historically important rearing habitats for many salmonid juveniles (Sedell and Luchessa 1982), and where these habitats remain intact they often contribute a disproportionate share of total salmonid abundance.

As detailed in Section 3.9.5, large woody debris interacts with natural channel-forming features such as boulders or bedrock to create different types of pool habitats (e.g., plunge pools, scour pools, eddy pools) and to increase hydraulic heterogeneity. The influence of large wood on both the formation of channel units and specific microhabitats creates habitat complexity that allows multiple species to coexist as an assemblage. in addition, large wood and associated pool habitats provide cover from predators and refuge habitats during storm events (Everest et al. 1985). Undercut banks and overhanging vegetation also serve as cover for juvenile anadromous and resident adult salmonids.

Substrate may also play an important role in habitat selection. in the summer months, boulders may provide both visual isolation from other fish and cover from predators. In winter, several salmonid species (e.g., steelhead, resident rainbow, and cutthroat trout as well as chinook salmon) have been observed to seek refuge in substrate interstices at low water temperatures (Chapman and Bjornn 1969; Bustard and Narver 1975; Campbell and Neuner 1985; Hillman et al. 1989). Gravel, cobble, and boulder substrates provide greater interstitial refugia than substrates dominated by sand or silt.

Flow and Depth

The amount of physical space available to juvenile and adult salmonids rearing in streams and the quality of that habitat is directly related to stream discharge (Everest et al. 1985). Within stream environments, salmonids select specific microhabitats where water depth and velocity fall within a specific range or where certain hydraulic properties occur (Table 5-5). These preferences in depth and velocity change both with season and life stage. Consequently, streamflow must be adequate to both satisfy minimum requirements for survival during periods of stress (e.g., low flow) as well as to provide specific microhabitat characteristics that are favorable to salmonid populations throughout their period of freshwater residence.

For many salmonids, smaller-sized fish tend to select shallower, slower moving waters than larger individuals (Chapman and Bjornn 1969; Everest and Chapman 1972; Moyle and Baltz 1985). Newly emerged fry may be vulnerable to downstream displacement by flow and typically select velocities lower than 10 cm/s (Bjornn and Reiser 1991). During summer months, salmonids often select

Table 5-5. Stream depths and velocities at holding sites of salmonids by age or size. From Bjornn and Reiser (1991). Reproduced with permission of the publisher.

Species and Source	Age* or Size	Depth (cm)	Velocity (cm/s)
Steelhead trout			
Bugert (1985)	31 – 44 mm	24	40
Everest and Chapman (1972)	0	< 15	< 15
	1	60 – 75	15 – 30
Hanson (1977)	1	51 mean	10 mean
	2	58 mean	15 mean
	3	60 mean	15 mean
Moyle and Baltz (1985)	0	35	7.3
	Juvenile	63	19.4
	Adult	82	28.6
Sheppard and Johnson (1985)	37 mm	< 30	< 25
Smith and Li (1983)	25 mm		4
	50 mm		8
	75 mm		18
	100 mm		24
	150 mm		24
Stuehrenberg (1975)	0	< 30	14 (range, 3 – 26)
	1	> 15	16 (range, 5 – 37)
Thompson (1972)	0	18 – 67	6 – 49
Chinook salmon			
Everest and Chapman (1972)	0	15 – 30	< 15
Konopacky (1984)	77 – 89 mm	55 – 60	12 – 30
			18 (dawn)
			12 (midday)
			25 (dusk)
Stuehrenberg (1975)	0	< 61	9 (range, 0 – 21)
	1	< 61	17 (range, 5 – 38)
Thompson (1972)	0	30 – 122	6 - 24
Steward and Bjornn (1987)	78 – 81 mm	40 – 58	8 - 10
Coho salmon			
Bugert (1985)	40 – 50 mm	24	39 (flume)
	0		15
	1		18
Nickelson and Reisenbichler (1977)	0	> 30	> 30
Pearson et al. (1970)	0		9 – 21
Sheppard and Johnson (1985)	62 mm	30 – 70	< 30
Thompson (1972)	0	30 – 122	5 – 24
Cutthroat trout			
Hanson (1977)	1	51 mean	10 mean
	2	56 mean	14 mean
	3	57 mean	20 mean
	4	54 mean	14 mean
Pratt (1984)	< 100 mm	32	10
	> 100 mm	62	22
Thompson (1972)	0, 1	40 – 122	6 – 49
Bull trout			
Pratt (1984)	< 100 mm	33	9
	> 100 mm	45	12

* Ages are in years or life stages, without units.

holding positions at moderate velocities but immediately adjacent to faster waters (Chapman and Bjornn 1969; Jenkins 1969; Everest and Chapman 1972). These positions are believed to confer the greatest energetic advantage to the fish. The amount of food delivered to a particular location is proportional to water velocity (Wankowski and Thorpe 1979; Smith and Li 1983). Consequently, fish that hold in water adjacent to faster feeding lanes can maximize food intake while minimizing energy expenditures associated with maintaining position in the current (Smith and Li 1983; Fausch 1984).

During winter months, metabolic demands and, thus, food requirements decrease as temperatures drop. Swimming ability also decreases with decreasing temperature (Brett 1971; Dickson and Kramer 1971; Griffiths and Alderdice 1972), and fish may be less able to maintain positions in fast waters for extended periods of time. **As** a result, salmonids tend to select slower water velocities, move to off-channel habitats, or seek refuge in substrate interstices when temperatures drop below a certain threshold temperature (Bustard and Narver 1975; Tschaplinski and Hartman 1983; Campbell and Neuner 1985; Johnson and Kucera 1985; Sheppard and Johnson 1985). Larger resident trout may abandon feeding sites in riffles and runs and move to slower-velocity pool habitats if substrate refugia are unavailable (Spence 1989).

For resident salmonids and juveniles of anadromous species that spend **a** year or more in freshwater, streamflow during the summer low-flow period must be adequate to prevent streams becoming excessively warm or drying up altogether. Under drought conditions, streams may become intermittent, and fish may be restricted to isolated pools. Such conditions can result in increased competition for food, reduced dissolved oxygen levels, increased physiological stress, and vulnerability to predators. Deep pools with groundwater inputs provide the necessary cover and thermal refugia.

Water Quality

Temperature. Juvenile and resident salmonids are variable in their temperature requirements, though most species are at risk when temperatures exceed 23–25°C (Bjornn and Reiser 1991). Upper and lower lethal temperatures as well as the "preferred" temperature ranges of several western salmonids is shown in Table 5-6. These values provide a general range of tolerable temperatures; however, the ability of fish to tolerate temperature extremes depends on their recent thermal history. Fish acclimated to low temperatures, for example, have lower temperature thresholds than those acclimated to warmer temperatures.

Table 5-6. Lower lethal, upper lethal, and preferred temperatures for selected salmonids. Based on techniques to determine incipient lethal temperatures (ILT) and critical thermal maxima (CTM). From Bjornn and Reiser (1991). Reproduced with permission of the publisher.

	Lethal temperature (°C)				
Species	Lower lethal*	Upper lethal†	Preferred temperature (°C)	Technique	Source
Chinook salmon	0.8	26.2	12–14	ILT	Brett (1952)
Coho salmon	1.7	26.0	12–14	ILT	Brett (1952)
		28.8‡		CTM	Becker and Genoway (1979)
Sockeye salmon	3.1	25.8	12–14	ILT	Brett (1952)
Chum salmon	0.5	25.4	12–14	ILT	Brett (1952)
Steelhead trout	0.0	23.9	10–13		Bell (1986)
Rainbow trout		29.4		CTM	Lee & Rinne (1980)
		25.0		ILT	Charlon et al. (1970)
Cutthroat trout	0.6	22.8			Bell (1986)

* Acclimation temperature was 10°C; no mortality occurred in 5,500 min.
† Acclimation temperature was 20°C unless noted otherwise; 50% mortality occurred in 1,000 min
‡ Acclimation temperature was 15°C.

Temperatures exceeding the upper "incipient lethal level" may be tolerated for brief periods, particularly during diel fluctuations, or may be avoided by seeking coldwater refugia provided by seeps or springs. Bull trout (not shown in table) appear particularly sensitive to warm waters. Temperatures higher than 14°C may act as a thermal barrier to migration of bull trout (OWRRI 1995). McPhail and Murray (1979) found that bull trout grew most rapidly at temperatures of 4°C, about 10°C colder than optimal growth temperatures for most species of *Oncorhynchus.* Lower lethal temperatures are near 0°C for most species of salmonids.

Many salmonid-bearing streams in the Pacific Northwest, particularly those in the southern, eastern, and low-elevation portions of the range, now experience maximum temperatures in summer that approach or exceed upper lethal levels for salmonids. Coldwater refugia in the form of springs, seeps, cold tributaries, and thermally stratified pools allow populations to persist in these streams that would otherwise be inhospitable. Nielsen et al. (1994) found that juvenile steelhead moved into thermally stratified pools when mainstem temperatures were between 23–28°C in a coastal northern California stream. Similarly, Li et al. (1991) reported that resident rainbow trout in an eastern Oregon stream selected natural and artificially created coldwater seep habitats when main-channel temperatures exceeded 24°C but showed no preference for coldwater areas when temperatures in the main channel dropped below 20°C.

Dissolved Oxygen. Salmonids are strong, active swimmers and require highly oxygenated waters. Maximum sustained swimming performance dropped off for coho and chinook salmon when DO concentrations decreased much below air-saturation levels (8–9 mg/L at 20°C) (Davis et al. 1963; Dahlberg et al. 1968). Alabaster et al. (1979) concluded that growth rate and food-conversion efficiency were probably limited if DO concentrations fell below 5 mg/L for Atlantic salmon. Davis (1975) estimated that salmonids would suffer no impairment if DO concentrations reniained near 8 mg/L and determined that DO deprivation would begin at approximately 6 mg/L. High water temperatures, which decrease oxygen solubility, further increase the stress on fish caused by low DO concentrations. A recent literature review resulted in criteria for salmonids presented in Table 5-7 (ODEQ 1995). The dissolved oxygen criteria developed for Idaho, Oregon, and Washington provide greater protection to salmonids than EPA's national minima.

Turbidity. Turbidity is elevated in all streams for short durations during storm and snowmelt events. Juveniles and adults appear to be little affected (Sorenson et al. 1977) by these transitory episodes, though Bisson and Bilby (1982) reported that coho salmon avoided water exceeding 70 nephelometric turbidity units (NTU), levels that may occur in some watersheds with high erosion potential. In a laboratory setting, juvenile coho salmon and steelhead trout exhibited reduced growth rates and higher emigration rates in turbid streams (25–50 NTU) compared to clear streams (Sigler et al. 1984). Lloyd et al. (1987) found that juvenile salmonids avoided chronically turbid streams including glacially influenced streams and those disturbed by human activities, Turbidity also influences foraging behavior of juvenile anadromous and adult resident salmonids by reducing the distance from which they can locate drifting prey.

5.2.4 Juvenile Migration

Depending upon the species or population, some juvenile salmonids migrate to the sea or to lakes, while others remain in a relatively small reach of stream for their entire lives (Everest et al. 1985; Bjornn and Reiser 1991). All species require unobstructed (either physically or chemically) access to upstream or downstream reaches for migration or dispersal to feeding grounds. In addition, species and stocks differ in their migratory behavior (i.e., timing and speed). For example, some species (e.g., pink and chum salmon) may move rapidly to the ocean over a few hours or days, while others (e.g., chinook salmon) may gradually move downstream over several weeks or months. These different behaviors entail substantially different habitat requirements during the migration period.

Physical Structure

Migrating fish are particularly vulnerable to predation because they often are concentrated and may move through areas with limited cover and high abundances of predators (Larsson 1985). Physical structure in the form of undercut banks and large woody debris provides refugia during resting periods and cover from predators. Juveniles that migrate to lakes, such as sockeye salmon or adfluvial resident populations, may be traveling upstream or downstream. In addition to cover from predators, these fish may require holding and feeding stations during their migrations. Artificial obstructions such as dams and diversions may impede migrations where they create unnatural hydraulic configurations.

Flow and Depth

Streamflow is important in facilitating downstream movement of salmonid smolts. Smolt migration is believed to be regulated by "priming" factors, such as photoperiod and temperature, that alter the

Table 5-7. Guidance for relating dissolved oxygen criteria to use protection. From ODEQ (1995).

Class	Concentration (mg/L) 30-day mean	7-day mean	7-day minimum	Minimum	Use/level of protection
Salmonid spawning		11		9 6*	Salmonid spawning and incubation of embryos. Low risk of impairment to aquatic community of salmonids, other native fish, and invertebrates.
Coldwater	**8**		6.5	6	Principally coldwater communities, salmon, trout, invertebrates, other native coolwater species througout all or most of the year. Juvenile anadromous salmonids may rear throughout the year. Low level risk of impairment for these groups.
Coolwater	6.5		5	**4**	Mixed native coolwater species, such as sculpins, and coolwater aquatic life. Provides migratory route for salmon and trout. Salmonids and other biota may be present during part or all of the year but may not dominate community structure. Slight level of risk to community.
Warmwater	5.5			**4**	Native warmwater fish; non-native species, salmonid migration; waterbodies may not naturally support native coolwater communities.
No risk		No change from natural			The only criteria that provides no additional risk to the resource is no change from background.

* Intragravel dissolved oxygen.

disposition of the fish in anticipation of downstream migration and "releasing" factors, including changes in temperature or streamflow, that trigger movement once a state of physiological "readiness" is obtained (Groot 1982). Dorn (1989) found that increases in streamflow triggered downstream movement of coho salmon in a western Washington stream. Similarly, Spence (1995) also found short-term increases in streamflow to be an important stimulus for smolt migration in four populations of coho salmon. Thus the normal range of streamflows may be required to maintain normal temporal patterns of migration.

Streamflow is also important in determining the rate at which smolts move downstream, although factors influencing the speed of migration remain poorly understood. Bjornn and Reiser (1991) state that the time required to travel from the Salmon River in Idaho to the Dalles Dam increased by as much as 30 days during low-flow periods following the construction of six dams on the Columbia-Snake system. In other systems, the migration speed of individuals may not be correlated to streamflow. This may occur in part because of the changing physiological disposition of fish during the run period with later migrants undergoing a more rapid smolt transformation as water warms.

Water Quality

Temperature. Temperature affects migration timing of smolts in two fundamental ways: by influencing the rate of growth and physiological development and by affecting the responsiveness of

fish to other environmental stimuli (Groot 1982). Consequently, alteration of thermal regimes through land-use practices and dam operations can influence the timing of migration. Holtby (1988) found that coho salmon smolts emigrated approximately 8 days earlier in response to logging-induced increases in stream temperatures. In addition, the age-class distribution was shifted from populations evenly split between one- and two-year old smolts to populations dominated by one-year old fish. A single year of poor ocean conditions will have a greater effect on a particular year class if the majority of smolts migrate at the same age, rather than spreading the risk over two years.

The specific temperature requirements of juvenile anadromous salmonids during their seaward migration are not well documented. Sockeye smolts have been reported to migrate at temperatures ranging from 2–10°C (Burgner 1991). Coho salmon have been observed to migrate at temperatures as low as 2.5°C and as high as 13.3°C (Sandercock 1991); however, most fish migrate before temperatures reach 11–12°C. Ocean-type chinook typically migrate during March and April at temperatures between 4.5°C and 15.5°C (Healey 1991), whereas stream-type chinook smolts tend to migrate 1 to 2 months later when conditions are substantially warmer. Once temperatures exceed a threshold level in the spring, salmon smolts will revert to a presmolt physiology and remain within the stream.

Dissolved Gases. Supersaturation of dissolved gasses (particularly nitrogen) has been found to cause gas bubble disease in upstream and downstream migrating salmonids (Ebel and Raymond 1976). Steelhead trout appear to be more susceptible than salmon to the disease because salmon have been found to better sense and avoid highly supersaturated waters (Stevens et al. 1980). However, all salmonids are susceptible. Although we found no information regarding dissolved oxygen requirements during seaward migration, it is likely that DO near saturation levels is required during this physiologically stressful period.

Turbidity. Turbid waters have been mentioned as affecting migration but little documentation is available in the literature. Thomas (1975) found fry migration increased as turbidity increased. Lloyd et al. (1987) found that turbid streams were avoided by juveniles except when the fish must pass through them along migration routes. There is also some evidence that diel migrations of salmonids is influenced by turbidity. Many salmonids tend to migrate during the evening hours (Burgner 1991), presumably to avoid predation. However, in streams with higher turbidity, migrations may be evenly dispersed during both the day and night.

6 Effects of Human Activities

Land-use practices—forestry, grazing, agriculture, urbanization, and mining—disrupt aquatic ecosystems by altering watershed processes that ultimately influence the attributes of streams, lakes, and estuaries. In this section, we review specific mechanisms by which human activities directly or indirectly affect aquatic ecosystems. With the exception of chemical contamination, most effects on watershed processes result from changes in vegetation and soil characteristics, which in turn affect the rate of delivery of water, sediments, nutrients, and other dissolved materials from uplands to stream channels. Within the riparian zone, land-use activities can alter the amount of solar radiation reaching the stream surface, affect the delivery of coarse and fine organic materials to streams, and modify fluvial processes that affect bank and channel stability, sediment transport, seasonal streamflow patterns, and flood dynamics. Disconnecting streams from their floodplains further alters hydrologic processes, nutrient dynamics, and vegetation characteristics.

Dams and water diversions affect salmonids by blocking or hindering migrations of fish, by altering the physical (e.g., temperature, flow, sediment routing) and chemical characteristics of streams, and by causing changes in stream biota. Other activities that influence salmonids and their habitats include wetland removal, harvesting of salmon, introduction of non-native species and hatchery salmonids, eradication of beaver, and activities associated with river, estuarine, and ocean traffic.

6.1 Forestry

Forest vegetation covers approximately 46% of the combined land surface of Washington, Oregon, and Idaho, including 34% of nonfederal lands and 58% of Federal lands (Pease 1993)'. Most commercial harvesting of timber is for softwoods, primarily Douglas-fir, ponderosa pine, hemlock, Sitka spruce, and lodgepole pine. Industry-owned lands, despite constituting only 18% of the softwood growing stock, accounted for 44% of the total softwood harvest in the region in 1986. In contrast, national and State forests constitute 70% of the growing stock, but provided only 46% of the softwood harvest (Jensen 1993). Intense production from nonfederal lands is likely to continue or increase as Federal timber supplies diminish.

Logging in the Pacific Northwest began in the mid 1800s, and by the 1860s the timber industry was well established. By 1880, forests along Puget Sound, as well as many rivers and streams, had been cleared for three or more kilometers inland (Sedell and Luchessa 1982). Throughout the 1900s forest harvest has continued, and the effects of logging have become pervasive across the region. Early forest practices were particularly damaging to stream environments. Splash damming was commonly used to float logs down to the sawmills, a practice that has had long-lasting effects on channel morphology and the abundance of large woody debris. Clear-cuts often included riparian forests, which yielded large quantities of wood that were easily transported downstream. Debris jams were routinely removed at the behest of biologists, who believed they hindered migration of anadromous fishes. Today the functional importance of large woody debris to salmonids is well documented (Bisson et al. 1987; Hicks et al. 1991a; Naiman et al. 1992) and State forest practice rules have been modified to reflect this knowledge. But despite recent improvements in forest practices, a legacy of past practices and cumulative effects will hamper our ability to quickly reverse habitat changes accrued from logging practices.

In the sections that follow, we review the effects of forest practices on watershed processes and salmonid habitats. We use the term "forest practices" to include all activities associated with the access, removal, and re-establishment of forest vegetation, including road construction, timber harvest, site preparation, planting, and intermediate treatments. Understanding the effects of these practices on natural processes will foster improved management, providing greater protection to salmonid habitats.

6.1.1 Effects on Vegetation

Forest practices directly influence vegetation within a watershed through the removal of trees during harvest, thinning, and road construction, and

'Percentages listed in Table 7 of Pease (1993) for nonfederal lands are in error. The correct total area for nonfederal lands is 33,616,655 hectares (83,066,500 acres). (P. Jackson, Geosciences, Oregon State University, personal communication.)

through manipulations of understory and ground vegetation designed to increase the vigor of desired species and inhibit growth of understory vegetation (e.g., burning or mechanical and chemical treatments). In addition, forest vegetation is indirectly affected by changes in site conditions following harvest. Removal of overstory vegetation can change local microclimate, soil moisture and stability, ground cover, and susceptibility to erosion, all of which influence the re-establishment of vegetation in the harvested area (Beschta et al. 1995). Soil compaction by ground-based equipment can reduce infiltration of water, thereby hindering the re-establishment of seedlings or the growth of established vegetation. In addition, as tree roots die after logging, subsurface spaces (macropores) become compacted or filled with sediment, reducing infiltration of water and reducing aeration in the soils. When porosity is reduced below 20%–25%, root growth is retarded. Mixing of mineral and organic soil layers also strongly influences the revegetation process (Beschta et al. 1995).

The magnitude of vegetation change and the succession of vegetation following logging depend on the type and degree of disturbance. For highly disturbed sites, early succession is dominated by colonizing annual and herbaceous species, followed by dominance/codominance of perennial species, and finally by dominance of overstory species. Where disturbance is less severe, residual species may dominate the early successional stages. Beschta et al. (1995) provide a more thorough review of the effects of forest practices on regeneration of vegetation.

At the landscape level, forest practices have resulted in substantial modification of species and age composition of western forests. Natural forests typically exhibit a mosaic of patches in different states of ecological succession. These mixed-age, multi-species plant assemblages have been replaced with even-aged forest plantations dominated by a single species. Riparian forests have been especially affected in areas where rapid growth of hardwood species (e.g., alder and maple) and shrubs (e.g., salmonberry) has precluded re-establishment of coniferous species (Bisson et al. 1987). In coastal streams, riparian areas outside of wilderness areas are dominated by alder and big leaf maple (FEMAT 1993). Certain conifers, such as western hemlock and Sitka spruce, frequently regenerate on partially decomposed nurse logs that are elevated above the forest floor (reviewed in Harmon et al. 1986). Removal of downed trees from the riparian zone may affect re-establishment of these species.

6.1.2 Effects on Soils

Forest practices can result in significant disturbance to soils, including increased compaction, scarification, and mixing of soil layers. The degree and effects of compaction are influenced by a number of factors, including the total area compacted, the soil type and moisture content, the equipment used, and the number of passes the vehicle makes over the site. Cafferata (1992 in Beschta et al. 1995) reviewed a number of studies and determined that 10%–40% of a harvest area may be compacted during tractor logging.

The effects of soil compaction appear to be of long duration. Studies have estimated recovery times from 10–50 years, with estimates as long as 90–110 years in an arid high-elevation site (Webb et al. 1986; Cafferata 1992). Duration of compaction depends upon depth of compaction, soil texture, soil temperature and moisture regimes, and biological activity. Recovery time increases with increasing depth of compaction. Soil recovery occurs more rapidly in clay soils that shrink and swell with changing moisture content, high elevation soils that are subjected to freezing and thawing, soils with high organic content that cushions them from compaction, and soils with high biological activity (e.g., burrowing rodents, earthworms, insects, soil microbes) (Beschta et al. 1995).

6.1.3 Effects on Hydrology

Timber harvest and its associated road construction and site preparation practices can have significant effects on hydrologic processes that determine streamflow. In most cases, the removal of vegetation increases the amount of water that infiltrates the soil and ultimately reaches the stream by reducing water losses from evapotranspiration. However, in forested systems where fog drip contributes significantly to total precipitation (Harr 1982), harvesting trees may have little effect on the total amount of water reaching the stream. Soil compaction can decrease infiltration and increase the likelihood of surface runoff. Roads can affect the routing of water by intercepting subsurface flow and diverting it down drainage ditches, effectively increasing drainage density within a watershed (Sidle et al. 1985). King and Tennyson (1984) observed altered hydrology when roads constituted 4% or more of catchment area (i.e., 4% was "roaded"). In snow-dominated systems, logging can influence the spatial distribution of snow on the ground, as well as the energy transfer processes that affect the melting rate of snowpack (Chamberlin et al. 1991). The effect of logging on hydrologic processes can change annual water yield, the magnitude and timing of peak flows, and the magnitude of summer low flows. The effects of logging on hydrologic processes are reviewed in two recent syntheses (Chamberlin et al. 1991; Beschta et al. 1995), and the material presented below is based primarily on these analyses.

Water Yield

In most instances, clear-cutting has been found to increase total water yield. In western Oregon, increases in water yield in the first 1–5 years following logging have ranged from a few centimeters to almost 65 cm (25 inches) (Beschta et al. 1995). The largest increases in yield occur in areas of high precipitation and high evapotranspiration. In these areas, increased yield resulted primarily from reduced evapotranspiration losses. One case study in the Bull Run watershed of Oregon found that water yields decreased slightly after logging (Harr 1982). Apparently, the decrease in evapotranspiration losses was offset by a decrease in the amount of fog-drip that normally occurred in the forested watershed.

In forests east of the Cascade Range, increases in water yield after logging are not as great. In a snow-dominated system in the Blue Mountains of Oregon, no increase in water yield was observed following logging (Fowler et al. 1987). The authors suggested that higher wind speeds after clear-cutting increase evaporation from snowpack, offsetting reductions in transpiration. Elsewhere in eastern Oregon and Washington, forested watersheds are likely to experience minor changes in total water yield. Beschta et al. (1995) speculated that in areas with low precipitation (< 30–51 cm [< 15–20 inches]) increases in water yield are likely to be negligible. Water yields in interior ponderosa pine and Douglas-fir forests are generally low to begin with because of high evaporation demands. Consequently, reductions in transpiration losses may be compensated by higher evaporation losses. In areas with higher precipitation, increases in water yield may be somewhat greater but still less than those observed in westside systems (Ziemer 1986; Beschta et al. 1995). Small increases in yield from snow-dominated systems in British Columbia and Colorado have also been reported (Hibbert 1967; Cheng 1989).

In addition to being affected by forest vegetation and climatic conditions, increases in water yield also depend on the percentage of the land area that is harvested or roaded (Hewlett and Nutter 1970; Trimble and Weirich 1987). In general, the increase in yield is directly proportional to the size of the area logged. However, for patch cuts, removal of vegetation may result in a smaller increased yield than predicted by area alone because of increased utilization of available moisture by vegetation in surrounding uncut areas (Beschta et al. 1995). For the same reason, selective harvesting or thinning may have minimal effect on water yield (Hibbert 1967). Bosch and Hewlett (1982) reviewed over 90 watershed studies and concluded that increased yield usually occurs after 20%–30% of a watershed has been harvested.

The effects of logging on total water yield persist until the transpiration demands of recovering vegetation approach those of uncut forests. In forests west of the Cascade Crest, return to natural conditions may take 30–40 years if no further disturbances occur in the watershed (Harr and Cundy 1992; Stednick and Kern 1992). Brush removal by mechanical means, chemical treatments, or burning (to aid re-establishing desired trees) can slow the rate of recovery. However, in general, these activities are practiced only until seedlings attain sufficient height to shade out competing species. Thus effects of these practices on water yield are likely to be short term.

Timing of Runoff

In rain-dominated systems, the largest increases in water yield because of logging generally occur during periods where both precipitation and transpiration rates of vegetation are relatively high, usually the fall (Chamberlin et al. 1991) and spring (Beschta et al. 1995). With reduced transpiration, soil moisture is rapidly replenished with the onset of rains in the fall and subsurface flow to stream channels commences (Rothacher 1971; Harr et al. 1979). Evapotranspiration losses from mature forests are comparatively small during the winter because of low temperatures and high humidity and, consequently, increased yield in winter is generally smaller (Chamberlin et al. 1991); however, in the spring, the differences between transpiration losses in mature forests and those in clear-cuts are again greater, and increases in water yield may be higher than in winter (Beschta et al. 1995).

In snow-dominated systems, increases in water yield generally occur during the early spring snowmelt period. The loss of shading following removal of the forest canopy can accelerate snowmelt, resulting in an earlier peak in the stream hydrograph. In snow-dominated systems, solar radiation is the primary factor influencing the rate of snowmelt (Chamberlin et al. 1991). In the snow-rain transition zone of the western Cascades snowmelt is driven primarily by convective transfer of sensible and latent heat to the snowpack (Harr 1986). Opening up the forest canopy can increase wind speed and turbulence, facilitating more rapid melting.

Peak Flows

A recent review (Beschta et al. 1995) of effects of timber harvest on peak flows in systems in the Pacific Northwest indicates a high degree of variability among sites. In rain-dominated systems of the Coast Range, most studies have indicated increases in peak flows following logging, particularly those occurring in fall (Table 6-1). In a few cases, increases have been insignificant, and in one case, a decrease in peak flows was observed.

Where increases in peak flow occur, they likely result from reduced evapotranspiration losses with removal of the forest canopy and more rapid routing of water to the stream channel because of soil compaction and roads.

In transient-snow systems of western Oregon responses of peak flows are similarly variable (Table 6-2). Several studies have indicated increases of 10%–200% in peak flows (Rothacher 1973; Harr et al. 1979), while others have shown no change or decreases (Harr et al. 1979; Harr et al. 1982; Harr and McCorison 1979). Harr (1986) re-analyzed published data and found that studies showing decreases in peak flows were inconclusive. In systems where harvest has increased peak flows in the transient-snow zone, it is believed that vegetation removal increased delivery of water to the soil from the snowpack during rain-on-snow events (Harr 1986). Coffin and Harr (1992) used lysimeters placed under the snowpack to confirm increased melt rates and delivery of water to the soil during rain-on-snow events. Maximum differences in melt rates between open and forested plots occurred when rain events were accompanied by relatively high temperatures and wind speeds, apparently because of increased transfer of sensible and latent heat to the snow. Coffin and Harr (1992) report that effects of harvesting on peak flows were still evident in 20–25 year-old plantations.

In snow-dominated systems, peak flows have generally shown little or no change following logging (Table 6-2) although studies are limited in geographic

Table 6-1. Effects of timber harvesting on peakflows in coastal areas of the Pacific Northwest. Modified from Beschta et al. (1995). Reproduced with permission from the principal author.

Watershedl Location	Size (acres)	Geology/Soils	Harvest method and percent	Peakflow effect	Reference
OREGON					
Needle Branch (Alsea watershed)	173	sandstone shallow GL-GCL*	cc† 82%	fall: +50% winter: +19%	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek–Main (Alsea watershed)	748	sandstone shallow GL-GCL	cc 26%	all: ns‡	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek–2 (Alsea watershed)	138	sandstone shallow GL-GCL	cc 90%	fall: +51% winter: +20%	Hsieh (1970) Harris (1977) Harr et al. (1975)
Deer Creek–3 (Alsea watershed)	99	sandstone shallow GL-GCL	cc 65%	fall: +50% winter: +30%	Hsieh (1970) Harris (1977) Harr et al. (1975)
BRITISH COLUMBIA					
West Coast–I	57	quartz GSL§	cc 71%	all: –22%	Cheng et al. (1975)
Carnation Creek B	2,470	volcanic coarse soil	cc 41%	all: ns	Hetherington (1987)
Carnation Creek	30	volcanic coarse soil	cc 90%	all: +20%	Hetherington (1987)
Jamieson Creek	739	na¶	cc 19.2%	winter: +13.5%	Golding (1987)
CALIFORNIA					
South Fork Caspar Creek	1,047	sandstone coarse soil	sc# 60%	small: +107% large: ns	Ziemer (1981)

* gravelly loam-gravelly clay loam (GL-GCL)
† clear-cut (cc)
‡ not significant (ns)
§ gravelly sandy loam (GSL)
¶ not available (na)
\# shelterwood (sc)

Table 6-2. Effects of timber harvesting on peakflows in interior areas of the Pacific Northwest. Modified from Beschta et al. (1995). Reproduced with permission of the principal author.

Watershed/ Location	Size (acres)	Geology/ soils	Harvest method and percent	Peakflow effects	Reference
RAIN-ON-SNOW					
Watershed—I* (OR)	235	basalt/GL†	cc* 100%	fall: +200% winter: ns§	Rothacher (1973)
Watershed—3 (OR)	249	basalt/GL	cc 25%	mean: +10%	Rothacher (1973)
Watershed—6 (OR)	32	basalt/GL	cc 100%	all: ns	Harr et al. (1982)
Watershed—7 (OR)	52	basalt/GL	sc¶ 60%	all: ns	Harr et al. (1982)
Watershed—10 (OR)	25	basalt/GL	cc 100%	ROS#: –36% rain: ns	Harr and McCorison (1979)
Coyote Creek—I** (OR)	170	basalt/ GL-GCL††	sc 50%	mean: +30% large: +48%	Harr et al. (1979)
Coyote Creek—2 (OR)	168	basalt/ GL-GCL	cc 30%	all: ns	Harr et al. (1979)
Coyote Creek—3 (OR)	121	basalt/ GL-GCL	cc 100%	mean: +44% large: +35%	Harr et al. (1979)
SNOWMELT					
High Ridge—I** (OR)	73	fractured basalt/SL§§	cc 43%	all: ns	Fowler et al. (1987)
High Ridge—2 (OR)	60	fractured basalt/SL	sc 50%	all: ns	Fowler et al. (1987)
High Ridge—4 (OR)	292	fractured basalt/SL	cc 22%	all: ns	Fowler et al. (1987)
Camp Creek (BC)¶¶	8,373	granite/ coarse soils	cc 30%	annual: –9 to +35%	Cheng (1989)

* Watersheds in H. J. Andrews Experimental Forest, western Cascades region
† gravelly loam (GL)
‡ clear-cut (cc)
§ not significant (ns)
¶ sheltetwood (sc)
\# rain-on-snow (ROS)
** Coyote Creek, South Umpqua Experimental Forest, Western Cascades Region
†† gravelly loam—gravelly clay loam (GL-GCL)
‡‡ High Ridge Evaluation Area, Blue Mountains Region
§§ silt loam (SL)
¶¶ Camp Creek, southern British Columbia, Canada

distribution. In most of these studies, the percentage of the watershed cut has been less than 50%. Despite the lack of conclusive data, it is reasonable to predict increased peak flows following logging. Snow accumulation is generally higher in open patches created by logging (Chamberlin et al. 1991), though it is unclear whether this is merely a redistribution of snow over the watershed or an actual increase in availability. Increased wind speeds in cleared areas may accelerate melting, leading to more rapid runoff and higher peak flows.

Low Flows

Increases in summer low flows have been observed following logging in a number of systems in the Pacific Northwest. Harr and Krygier (1972) documented average increases in summer flows of 60% following logging of a Coast Range stream in

Oregon. Somewhat larger increases were observed in a western Cascade stream (Rothacher 1970). Keppeler and Ziemer (1990) noted increases in summer flows in a northern California stream, but found that the increases disappeared within five years. Studies in drier, interior climates have been less conclusive. Cheng (1989) reported increases in summer streamflows that persisted for six years after logging of a basin in the interior of British Columbia. However, Troendle (1983) found no increase in summer low flows after logging in Colorado. Where increases in summer flows occur, they likely result from reductions in evapotranspiration losses.

Few long-term studies of effects of logging on low flows have been performed. A notable exception is Hicks et al. (1991b) who found that August streamflows in a central Oregon Cascade stream increased for 8 years following logging, but decreased for 18 of the next 19 years. On average, August streamflows were 25% lower than in prelogging years. The authors attributed reductions in streamflow to the replacement of coniferous vegetation with more water-consumptive hardwood species. Thus, the long-term effects of logging on streamflows likely depend on vegetation composition before and after harvest,

6.1.4 Effects on Sediment Transport

Forest practices can substantially increase delivery of sediments to streams through both surface erosion and mass wasting. The effect of forest practices on sediment transport depends on a number of local site conditions including climate, vegetation, topography, and soil type, as well as on specific aspects of the activity, including the type and areal extent of disturbance and the proximity of the disturbance to the stream channel. Thus, the relative effects of roading, timber harvest, site preparation, and other forest practices on sediment production vary with location (Beschta et al. 1995).

Furniss et al. (1991) concluded that forest roads contributed more sediment than all other forest activities combined on a per unit area basis. Summarizing results from nine different studies, they reported that mass wasting associated with roads produced 26–346 times the volume of sediment as undisturbed forests. Mass failures were attributed to poor road location, construction, and maintenance, as well as inadequate culverts. Beschta (1978) found that, in three out of eight years, suspended sediment increased significantly from two catchments in the Coast Range, primarily as a result of mass failures from roads. Mass failures associated with roads most commonly occur on cut and fill slopes, but may also begin where end-haul material is deposited on a hillslope (Dent **1993).** In addition, channel constrictions at road crossings may lead to bank sloughing and bank erosion.

Surface erosion from roading also constitutes a significant source of chronic sediment inputs (Beschta et al. 1995). Splash erosion mobilizes sediment on exposed road surfaces, and runoff from compacted surfaces may also facilitate sediment transport. Water diverted into ditches along roads gathers energy and can cause significant erosion at the outlets of cross-drain culverts (Beschta et al. 1995). Montgomery (1993), studying three small catchments, found that roads decreased the drainage area needed to support a channel head and thereby increased the length of the channel. He attributed this phenomenon to lower infiltration or greater runoff on roads. He also demonstrated that channel expansion was a function of catchment area. The combined effects of mass wasting and surface erosion can lead to elevated sediment levels in streams even when only a small percentage of a watershed is roaded. For example, Cederholm et al. (1981) reported increased sediments in salmonid spawning gravels when roads exceeded **3%** of the total basin area.

Sediment delivery from other forest activities, including harvest, yarding, and site preparation, may be increased via several mechanisms. Loss of the protective vegetative cover can increase splash erosion and decrease slope stability (Swanston et al. 1980; Marcus et al. 1990). Yarding activities cause extensive soil disturbance and compaction which may increase splash erosion and channelized runoff. Ground-based vehicles moving logs from felled trees and skidding logs to landing sites compact and scarify the soil. Compaction of the decomposing root systems reduces the infiltration capacity of these channels, leading to slumps, landslides, and surface erosion (Everest et al. **1987).** Loss of the humic layer through mechanical disturbance and fire further increases the potential for surface erosion.

The quantity of sediments delivered to the stream channel depends upon the integrity of the riparian zone, the intensity of disturbance, the areal extent of the disturbance, the proximity of the disturbance to the stream channel, and slope steepness. Site disturbance may be intensified by a hot bum following harvest which creates extensive areas of bare soil (Everest et al. 1987). Piling and burning versus broadcast burning will also intensify the site disturbance and increase sediment delivery rates. Riparian buffer strips and buffer strips below roads can trap sediments, significantly reducing the delivery rate (Swanston 1991).

The fate of sediments once in the channel also depends on the nature of sediments (coarse versus fine) and local site characteristics. Although surface erosion is less dramatic and less evident than mass wasting, it may be more detrimental to stream biota because the delivery of particles occurs over a longer time, and those particles are smaller and more likely to become embedded in coarser substrates. Bilby (1985) found that sediments from road crossings were flushed from a fourth-order stream reach of 2% gradient that drained a small (5.5 km^2) catchment with a relief ratio of 0.10. Presumably, these sediments (2–151 $mg \cdot L^{-1}$ above control levels) were

Table 6-3. Summary of summer temperature changes associated with management activities on forested watersheds in the Pacific Northwest. From Beschta et ai. (1987). Reproduced with permission from the publisher.

Location	Treatment	Stream temperature variables	Temperature change (°C)	Reference
Alaska (Southeast)	Clear-cut and natural openings	ATemperature per 100 m of channel	+0.1 – 1.1°C per 100 m; Avg = 0.7°C per 100 m	Meehan (1970)
British Columbia (Vancouver Island)	Logged (Tributary H)	Average Jun–Aug diurnal temperature range	+0.5 – 1.8°C over pretreatment levels	Holtby and Newcombe (1982)
	Logged and burned (Tributary J)	Average Jun–Aug diurnal temperature range	+0.7 – 3.2°C over pretreatment levels	Holtby and Newcombe (1982)
Oregon (Cascades)	Clear-cut	Average Jun–Aug maximum	+4.4 – 6.7°C	Levno and Rothacher (1967)
	Clear-cut and burning	Average Jun–Aug maximum	+6.7 – 7.8°C	Levno and Rothacher (1967)
Oregon (Coast Range)	Clear-cut	Average Jul–Sep maximum	+2.8 – 7.8°C	Brown and Krygier (1967)
	Clear-cut and burning	Average Jul–Aug maximum	+9 – 10°C	Brown and Krygier (1970)
Oregon (Cascades)	Mixed clear-cut and forested reaches	A Temperature per 100 m of channel	+0 – 0.7°C per 100 m	Brown et al. (1971)
	Tractor stripped area	A Temperature per 100 m of channel	+15.8°C per 100 m	Brown et al. (1971)

deposited downstream. In a separate study, Duncan et al. (1987) reported that first- or second-order channels with high amounts of wood debris retained 55% of road-crossing sediments at flows up to 7% of bankfull. Thus stream gradient and retentive in-channel structures appear important in determining whether sediments are deposited locally or transported downstream.

6.1.5 Effects on Thermal Energy Transfer and Stream Temperature

Logging most directly affects energy transfer by reducing shade provided by riparian vegetation, which increases the amount of direct solar radiation reaching the stream surface. The increase in energy reaching the stream depends on the amount of shading lost. Measurements from an old-growth Douglas fir forest in western Oregon indicated shading averaged 84% (Summers 1983). Brazier and Brown (1973) reported that angular canopy densities generally fall between 80% and 90% in old-growth stands in western Washington (cited in Beschta et al. 1987). In eastern Oregon, natural canopy density is somewhat less. Slightly lower shading (75%) has been reported for a stream in northern California (Erman et al. 1977), and Anderson et al. (1993) estimated shading in old-growth forests of the Upper Grande Ronde basin in eastern Oregon to be around 72%. Thus, the magnitude of increase in stream temperatures following canopy removal is likely to differ across the region (Table 6-3).

Removal **of** riparian canopy also **affects** other energy transfer processes including convection, evaporation, and advection. Convective and evaporative heat exchange are both affected by wind speed (see Section 3.7), which generally increases as

riparian vegetation is removed. Consequently, convective exchange as well as evaporative losses tend to increase slightly following logging (Brown 1969). The removal of vegetation from upslope areas generally allows greater heating of the soil surface during the summer months. Rain falling in the early part of the rainy season may pick up additional heat as it passes through the soil and infiltrates subsurface aquifers, resulting in increases in ground water temperature after logging.

Removal of riparian canopy has been shown to have two major effects on temperatures of smaller streams in the Pacific Northwest: increased maximum temperatures (particularly in summer) and increased diel fluctuations (Beschta et al. 1987; Beschta et al. 1995). For coniferous forests of the Coast Range and western Cascades, increases in average summer maximum temperatures because of clear-cutting have ranged from about 3 to 8°C (Table 6-3); (Beschta et al. 1987). Increases up to 10°C have been observed when clear-cutting has been followed by slash burning (Brown and Krygier 1970). Increases in annual maximum daily temperatures can be substantially greater. Hall and Lantz (1969) reported increases in maximum daily stream temperatures of up to 16°C in years immediately following logging of a small watershed in the Coast Range of Oregon. Holtby (1988) reported that average monthly water temperatures increased from 0.7 to 3.2°C following logging of the Carnation Creek (British Columbia) watershed, with the largest increases occurring in May–September and the smallest increases in December and January. These changes persisted for at least seven years after logging. Average diel temperature fluctuations increased by as much as 3.7°C in two Carnation Creek tributaries that had diel fluctuations of less than 1°C prior to logging (Holtby and Newcombe 1982). Hall and Lantz (1969) reported that midsummer diel fluctuations of 15°C were common in Needle Branch, Oregon, after logging. Documentation of temperature changes resulting from logging east of the Cascade range is sparse. Because the degree of shading provided by more open forest types (*e.g.*, ponderosa pine) is lower than for coastal and western Cascade streams, the increase in temperatures resulting from canopy removal might be expected to be slightly less. Nevertheless, because many streams east of the Cascades approach the maximum thermal tolerance level for salmonids during the summer, smaller increases in temperature might be equally or more detrimental to salmonids.

Although summer stream temperatures have been the focus of most research on the effects of logging on stream temperatures, changes in winter stream temperatures may also occur. Theoretically, the loss of riparian vegetation allows for greater radiative cooling at night during the winter months, potentially decreasing winter temperatures. However, Holtby (1988) reported increases in February–April mean temperatures of 1–2°C. Increases in groundwater temperatures following canopy removal may have been responsible for the increase in winter temperatures. Hall and Lantz (1969) also noted similar increases in temperatures during the winter in a coastal Oregon stream after the entire basin was clear-cut.

The magnitude of temperature change following removal of riparian vegetation depends on the size of stream and channel morphology. Because stream discharge and depth increase downstream, the ability of solar radiation to effect stream temperatures also diminishes with increasing stream size (Beschta et al. 1995). Moreover, the amount of shading provided by riparian vegetation decreases as streams become larger and wider. Consequently, the removal of riparian vegetation effects temperature most in small- and medium-sized streams, and least in large river systems. Sullivan and Adams (1990) suggest that riparian vegetation has a negligible effect on stream temperatures for streams that are 5th order or larger.

Although the effects of logging on stream temperatures within the logged area are well documented, the cumulative effects of temperature increases both downstream and over time are less well understood. Temperature data from Needle Branch in Oregon's Coast Range indicate that thermal regimes returned to near normal approximately seven years after logging and slash burning (Hall et al. 1987). In this case, alder replaced conifers as the dominant riparian vegetation and provided significant shade to this small stream. However, temperature increases in Carnation Creek showed no sign of diminishing eight years after logging and the author estimated that elevated stream temperatures were likely to persist for an additional decade or more (Holtby 1988). Similarly, in the higher elevation fir zone of the Cascade Range, the degree of shading may not reach prelogging levels for 40 years or more (Summers 1983). Thus the duration of temperature effects depends on the rate of recovery of riparian vegetation and the level of shading provided.

The cumulative effects of stream temperature changes downstream of logged areas are not well documented. As streams leave harvested areas and re-enter forested reaches, temperatures tend to decline as solar radiation is reduced. Similarly, small tributaries generally have a minor affect on the temperatures of larger streams which they enter (Caldwell et al. 1991). Sullivan and Adams (1990) have argued that in streams in western Washington, temperatures approach mean air temperatures at a "threshold distance" downstream from the watershed

divide. The cumulative effects of warming of upstream tributaries may have little affect beyond this distance, though no analysis has been conducted to validate this hypothesis outside of western Washington (Beschta et al. 1995). In a study in the western Cascades of Oregon, Beschta and Taylor (1988) found that stream temperatures increased with increased logging and road building in the basin. As logging activity decreased in subsequent years, temperatures also declined, strongly suggesting that the cumulative effects of logging and road building were responsible for the previous temperature increases.

6.1.6 Effects on Nutrients

Forest practices can lead to changes in nutrient distribution and dynamics in upland areas, which in turn affect availability in streams. Nutrients are directly lost to the ecosystem through the removal of trees. Harvest intensity (i.e., proportion of forest canopy removed), type of harvest (logs or whole tree) and cutting frequency all affect the rate of nutrient removal from the system (Beschta et al. 1995). Despite the loss of nutrients stored in removed biomass, nutrients are generally more available to stream organisms in the years immediately following harvest (Figure 6-1). This results in part from the addition of slash to the forest floor (Frazer et al. 1990), accelerated decomposition of organic litter resulting from increased sunlight reaching the ground (Beschta et al. 1995), increased water availability for leaching of materials, and increased overland runoff and erosion that contributes unbound (nitrate and ammonium) and bound(orthophosphate) nutrients to the stream (Gregory et al. 1987). Where logging reduces riparian vegetation, nutrient supply to the stream (e.g., leaf litter and woody debris) may be reduced. As soils stabilize and revegetation occurs, the nutrient flux declines, though nutrients from herbaceous plants in the riparian zone add high quality materials that easily decompose. Over time herbs, shrubs, deciduous trees, and conifers provide allochthonous inputs for nutrient uptake (Figure 6-1).

Burning of slash, or the entire harvested area, can temporarily elevate the concentrations of nutrients entering the stream. Grier et al. (1989) suggests that fire effectively accelerates decomposition processes. If a fire is hot, however, much of the nitrogen is volatilized and lost to the system (Gessel and Cole 1973). In another study, potassium, phosphorus, calcium and magnesium increased by 2–8 times, while nitrogen decreased by two thirds following burning (Austin and Baisinger 1955). Herbicide treatments, like burning, can lead *to* short-term increases in nutrients as deciduous vegetation dies and decomposes.

The significance of forest harvest on nutrient losses depends on the mechanism causing the loss. The most significant losses result from tree removal. Leaching is not considered a major component of losses overall, accounting for less than 1% of losses from harvest. Losses because of volatilization resulting from fire can be much more significant (Beschta et al. 1995).

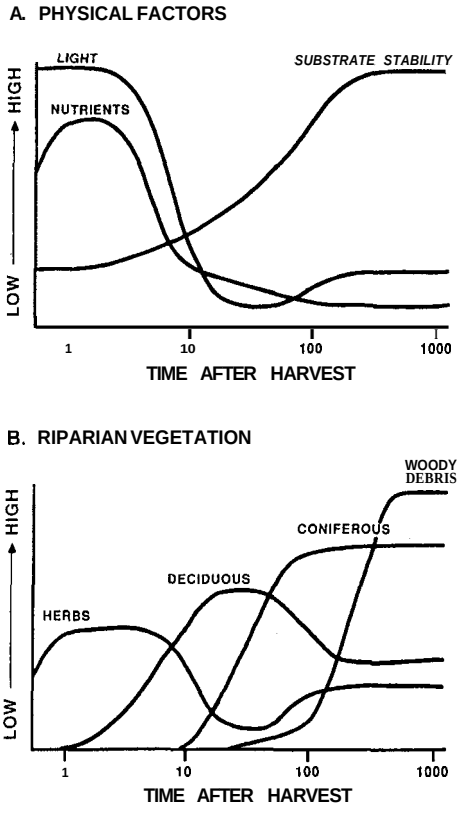


Figure 6-1. Temporal patterns of physical factors (A) and riparian vegetation (B) after timber harvest (time is expressed as years on a logarithmic scale.) From Gregory et al. (1987). Reproduced with permission from the publisher.

6.1.7 Effects of Forest Chemicals

Fertilizers, herbicides, and insecticides are commonly used in forest environments to prepare sites for planting, to release and stimulate growth of conifers, and to control diseases and pests. In addition, fire retardants are used to halt the spread of wildfire on forest lands. All of these chemicals can affect salmonids through several direct and indirect pathways. Fertilizers, pesticides, and fire retardants that reach surface waters can be toxic to salmonids or may alter primary and secondary production, influencing the amount and type of food available to salmonids, Fertilizers and pesticides indirectly affect

salmonids by influencing the rate of recovery of upland and riparian vegetation following logging, which in turn affects hydrologic processes, delivery of sediment and organic debris, heat transfer, nutrient cycling, and soil biota.

Contamination of surface waters by forest chemicals and the resultant risk of toxic effects on aquatic organisms depend on several factors, including the form and application rate of the chemical (and associated carriers), the application method (i.e., aerial versus ground spraying), soil type, weather conditions during and after application, and the retention of no-spray buffers in the riparian zone. The specific formulation determines the solubility of the chemical in water and its tendency to adsorb onto organic or inorganic matter within the soil, both of which determine the substance's mobility within the soil column. The persistence of forest chemicals in the environment varies. Some chemicals are highly volatile or are rapidly broken down through microbial activity or photodecomposition, whereas others persist for months or years. The volatility and adsorption of chemicals to soil particles are affected by temperature and soil moisture. If chemical application is followed by precipitation events, the likelihood that chemicals will reach surface waters increases. Contamination of surface waters also occurs when chemicals that have been applied to ephemeral channels are later mobilized during rainstorms. Aquatic organisms may be exposed to forest chemicals through direct contact with contaminated water, sediments, or food. No-spray buffers around streams (including ephemeral streams) and riparian areas substantially reduce the risk of contamination (Norris et al. 1991); however, toxic levels of chemicals may reach streams from storm runoff and wind drift even when best management practices are employed (e.g., Rashin and Graber 1993). Indirect effects of chemicals on watershed processes are largely unavoidable because change in vegetation is usually the desired outcome of such applications. Although both direct and indirect effects of forest chemicals on salmonids may be significant, it is important to note that less than 1% of total pesticides applied in the United States are used in forestry (Norris et al. 1991); thus, contamination from forest practices is likely minor compared to contamination from agricultural practices.

Fertilizers

Fertilizers are used in forest settings to replace nutrients lost during and after timber harvest and to accelerate growth of conifers. Application of fertilizers to a catchment typically results in increased concentrations of nutrients, particularly nitrogen, in streams. A recent review of effects of forest fertilization on water quality and aquatic biota indicates that urea application typically leads to elevated levels of urea-N, ammonia-N, and nitrate-N in surface waters (Bisson et al. 1992a). Urea-N usually dissipates within a few days, whereas ammonia-N may be elevated for months and nitrate-N for a year or more. The concentration of nitrogen within the stream depends on a number of factors, including the percentage of the watershed fertilized, the application rate, the drainage density (stream km relative to total watershed area), the width of unfertilized buffers along streams, and whether or not precipitation occurs following application. Although drinking water and aquatic standards are typically not exceeded with most applications, the elevation of nitrogen has the potential to promote growth of periphyton, which in turn may influence production of invertebrates and fishes. However, Bisson et al. (1992a) concluded that enhanced fish production because of forest fertilization has not been demonstrated in the Pacific Northwest. An indirect benefit of fertilizer applications is more rapid growth of vegetation within the catchment, which in turn accelerates the recovery of natural hydrologic regimes and sediment delivery rates.

Herbicides

In forest plantations, a wide variety of herbicides are used to control the invading hardwoods, herbaceous plants, and grasses to enhance the suitability of the area for re-establishment of desired tree species. Ten herbicides commonly used in forestry are 2,4-D, picloram, hexazinone, atrazine, imazapyr, triclopyr, forsamine, glyphosate, dalapon, and dinoseb. The behavior and toxicity of these substances is reviewed in detail in Norris et al. (1991) and Beschta et al. (1995), from whom much of the information below was excerpted.

The risk of toxicological effects of herbicides on salmonids is greatest when herbicides are directly applied to surface waters or reach surface waters by wind drift. Whether herbicides applied to upland forests will reach surface waters depends on their volatility, mobility in the soil, and persistence in the environment. Of the herbicides commonly used in forest applications, hexazinone, atrazine, imazapyr, and triclopyr are generally the most persistent, with soil half-lives of 2–6 months or more, depending on soil type. The half-lives of most other forest herbicides are generally from 2–5 weeks. Although there is substantial literature on the toxicity of various herbicides to salmonids, most of the available information comes from laboratory studies rather than the field. These laboratory studies focus on acute lethal doses (Reid 1993). Sublethal effects of herbicides on salmonids include reduced growth, decreased reproductive success, altered behavior, and

reduced resistance to stress (reviewed in Beschta et al. 1995). Sublethal exposures of picloram were found to increase mortality by 70% in yearling coho exposed to seawater (Lorz et al. 1979). Information on effects of herbicides in aquatic invertebrates is also scarce. Hartman and Scrivener (1990) reported a **42%** reduction in the density of aquatic macroinvertebrates for 1.5 years following application of Roundup. These reductions were attributed to herbicide-induced irritation and drift of invertebrates coupled with high flows and decreases in substrate stability.

Herbicides used to release conifers from competing vegetation can accelerate the long-term recovery of upland and riparian areas. Over the short term, the elimination of deciduous vegetation can affect streams in several ways, both positive and negative. Herbicide applications in upland areas slow the recovery of vegetation, prolonging disruption to hydrologic and sediment delivery processes. Within the riparian zone, removal of deciduous vegetation increases solar radiation reaching streams, which stimulates algal production, potentially increasing the food base for invertebrates and fish. Delayed production of deciduous trees and accelerated growth of conifers reduces the delivery of leaves and intermediate-sized wood to streams over the short term, but increases the potential for recruitment of large coniferous wood over longer periods. Depending on whether temperature, spawning sites, cover, or food is limiting, these changes may initially hinder or aid salmonid production.

Insecticides

Insecticides are used both to prevent insect infestations and to control insect outbreaks once they have occurred. In general, insecticides are more toxic to fish and other aquatic biota than herbicides; however, they usually are applied at lower rates (Beschta et al. 1995). The greatest effect of insecticide on fish probably arises from effects on terrestrial and aquatic insects that form the salmonids' food base. Forest insecticides cause direct mortality to these insects or may stimulate catastrophic drift of aquatic invertebrates out of the affected stream reach. In addition, benthic algal communities in streams are frequently controlled by grazing invertebrates; consequently, the loss of invertebrates may release primary production, causing fundamental shifts in the trophic structure of streams. Norris et al. (1991) concluded that insecticides generally have shorter term effects on stream ecosystems than herbicides but that the effects may be more dramatic. Populations of invertebrates may take months to recover following insecticide applications, and full recovery of the invertebrate assemblage may take several years (Norris et al. 1991). Because salmonids in some forest streams may be food-limited, reductions in aquatic insect biomass and altered assemblage composition may result in reduced growth and numbers of salmonids. For example, Kingsbury (1983 in Norris et al. 1991) reported a decline in the growth rate of Atlantic salmon parr immediately following treatment with an insecticide; however, by the end of summer, fish in treated and untreated reaches were of similar size. Direct toxic effects may occur if salmonids consume drifting, pesticide-laden, aquatic organisms or terrestrial insects that fall into streams. Other indirect effects of insecticides on salmonid habitats are not well documented; however, protection of trees from insect pests may reduce the number of trees that die and fall into streams, thereby reducing recruitment of large woody debris.

Fire Retardants

The use of chemical fire retardants plays and important role in the suppression of wildfires in the west. Historically, a variety of chemicals have been used to suppress fires; however, ammonium-based retardants account for nearly all chemical retardants used today (Norris and Webb 1989). Although documentation of adverse effects of fire retardants on salmonids is scarce, quantities of retardant dropped during fires may be significant, and cases of fish mortality caused by retardants have been reported. For example, approximately 5.3 million liters of retardant were used to fight the Yellowstone fire of 1988, and at least two small fish kills (approximately 100 fish each) were reported (Schullery 1989). Fire retardant killed approximately 700 adult salmon, as well as a large number of juveniles, in an Alaskan stream (Hakala et al. 1971 in Norris and Webb 1989). Potential indirect effects of fire retardants on salmonids include mortality of invertebrates and eutrophication of downstream reaches (from phosphates). The extent of effects of retardants on aquatic ecosystems is influenced by application procedures (quantity applied, line of flight of aircraft relative to the stream), site characteristics (stream width-depth ratio, degree of canopy cover), and streamflow.

6.1.8 Effects on Physical Habitat Structure

Timber management activities have resulted in substantial modification of the physical characteristics of stream habitats throughout forested regions of the Pacific Northwest. Many of these changes have resulted from decreased recruitment of large woody debris (LWD) from the riparian zone and intentional removal of LWD from stream channels (Bisson et al. 1987; Maser et al. 1988; Hicks et al. 1991a). Removal of vegetation from the riparian zone has

altered sources, mechanisms for delivery, distribution patterns, and stability of wood in stream channels (Bisson et al. 1987). Hardwoods have replaced conifers in many riparian areas following logging; woody debris produced by deciduous vegetation tends to be smaller, more mobile, and shorter-lived than that derived from conifers and, consequently, does not function as well in retaining sediment. The reduced supply of large woody debris decreases channel stability and eventually leads to loss of instream cover and pool habitat available for fish (Bisson et al. 1987). During the winter, salmonids have been reported to abandon reaches that are devoid of large wood (Tschaplinski and Hartman 1983). In addition, loss of large woody debris results in decreased retention of sediments, including gravels used by salmonids for spawning, as well as organic materials. The lack of debris also simplifies channel hydraulics, diminishing the heterogeneity that allows fishes to segregate among microhabitats. Loss of riparian vegetation also leaves banks unprotected, increasing bank erosion and reducing the formation of undercut banks that salmonids frequently use for cover. Hicks et al. (1991a) provide a good summary of short and long-term physical effects of forest practices on stream habitats (Table 6-4).

Excessive sedimentation resulting from logging and associated roads has also played a substantial role in altering salmonids habitats. Several studies have recorded increased levels of fine sediment in spawning gravels following logging (Cederholm and Reid 1987; Hall et al. 1987; Hartman et al. 1987); reductions in production of salmonids have been attributed in part to sedimentation of redds in each of theses instances, Fine sediments generated by logging and roading activities also can fill substrate interstices, decreasing the availability of cover for juvenile salmonids and altering primary production and invertebrate abundance (Cederholm and Reid 1987; Hicks et al. 1991a). Inputs of coarse sediments can fill pool habitats, resulting in channel shallowing and widening, and decreasing channel stability.

Culverts on logging roads create physical obstructions that fish must negotiate when migrating to and from the ocean and between summer or winter rearing habitats. Poorly designed and installed culverts act as barriers to both anadromous and resident salmonids. At culverts, excessive flow velocities, insufficient water, excessive culvert heights, and the absence of pools all can impede migration (Evans and Johnson 1980). Culverts also fail frequently when inappropriately designed and installed, resulting in mass wasting of road crossings. Road construction along streams, particularly where revetments are required, can constrain streamflow, thereby facilitating scouring of the channel bed. Dose and Roper (1994) found that channels widened as road density in catchments increased.

The cumulative effect of forest practices has been a reduction in the complexity of stream macro- and microhabitats. FEMAT (1993) documents substantial decreases in the number of large, deep pools in river systems west of the Cascade Range, which were attributed to loss of pool-forming structures (e.g., boulders, large wood), filling of pools with sediment, and loss of sinuosity of stream channels. Similar declines in pool frequency in river basins of eastern Oregon and Washington are reported in McIntosh et al. (1994a, 1994b). Reductions in pool habitat arise from the combined effects of loss of large woody debris, increased sediment inputs (which fill pools), and hydrologic changes that accompany land use. Although logging-related activities are not the only cause of these changes, they certainly play an important role. Effects of logging on physical habitat structure may persist for decades to a century or more. However, the causal linkages between land use, habitat development, and fish impacts are not always clear.

6.1.9 Effects on Stream Biota

A substantial volume of literature documents the effects of logging on aquatic biota (see reviews in Gregory et al. 1987; Hicks et al. 1991a; and Beschta et al. 1995). The response of aquatic communities to logging depends on a variety of factors, thus studies have sometimes produced seemingly contradictory results. Any of the following factors may influence the specific response of a given system: 1) species and stocks of fishes are diverse and adapted to local conditions; thus, the response may vary in different portions of each species' range; 2) physical and vegetative conditions, as well as logging methods, vary among regions such that impacts differ in magnitude, persistence, and ecological significance; 3) biotic interactions and long-range fish movements can mediate the effects of habitat alteration, such that most important biotic changes are indirectly and incompletely related to physical effects; 4) impacts of numerous independent factors can accumulate over time or space, or interact in either a compensatory or synergistic way, making ecological responses complex and difficult to predict; 5) dynamic, sometimes catastrophic natural events (e.g., large floods, changing oceanic conditions) create variable "baseline" conditions making it difficult to quantify additional variability caused by habitat alterations (Frissell 1991).

Within this context, some common patterns in the response of aquatic organisms to forest practices have been identified, based on both empirical evidence and theoretical expectations (e.g., the river continuum concept). In smaller streams, the removal of riparian

Table 6-4. Influences of timber harvest on physical characteristics of stream environments, potential changes in habitat quality, and resultant consequences for salmonid growth and survival. From Hicks et al. (1991a). Reproduced with permission from the publisher.

Forest Practice	Potential change in physical stream environment	Potential change in quality of salmonid habitat	Potential consequences for salmonid growth and survival
Timber harvest from streamside areas	Increased incident solar radiation	Increased stream temperature; higher light levels; increased autotrophic production	Reduced growth efficiency; increased susceptibility to disease; increased food production; changes in growth rate and age at smoking
	Decreased supply of large woody debris	Reduced cover; loss of pool habitat; reduced protection from peak flows; reduced storage of gravel and organic matter; loss of hydraulic complexity	Increased vulnerability to predation; lower winter survival; reduced carrying capacity; less spawning gravel; reduced food production; loss of species diversity
	Addition of logging slash (needles, bark, branches)	Short-term increase in dissolved oxygen demand; increased amount of fine particulate organic matter; increased cover	Reduced spawning success: short-term increase in food production; increased survival of juveniles
	Erosion of streambanks	Loss of cover along edge of channel; increased stream width; reduced depth	Increased vulnerability to predation; increased carrying capacity for age-0 fish, but reduced carrying capacity for age-1 and older fish
		Increased fine sediment in spawning gravels and food production areas	Reduced spawning success; reduced food supply
Timber harvest from hillslopes; forest roads	Altered streamflow regime	Short-term increase in streamflows during summer	Short-term increase in survival
		Increased severity of some peak flow events	Embryo mortality caused by bed-load movement
	Accelerated surface erosion and mass wasting	Increased fine sediment in stream gravels	Reduced spawning success: reduced food abundance; loss of winter hiding space
		Increased supply of coarse sediment	Increased or decreased rearing capacity
		Increased frequency of debris torrents; loss of instream cover in the torrent track; improved cover in some debris jams	Blockage to migrations; reduced survival in the torrent track; improved winter habitat in some torrent deposits
	Increased nutrient runoff	Elevated nutrient levels in streams	Increased food production
	Increased number of road crossings	Physical obstructions in stream channel; input of fine sediment from road surfaces	Restriction of upstream movement; reduced feeding efficiency
Scarification and slash burning (preparation of soil for reforestation)	Increased nutrient runoff	Short-term elevation of nutrient levels in streams	Temporary increase in food production
	Inputs of fine inorganic and organic matter	Increased fine sediment in spawning gravels and food production areas; short-term increase in dissolved oxygen demand	Reduced spawning success

vegetation increases light intensity, which stimulates the growth of benthic algae (Gregory 1980; Murphy et al. 1981; Shortreed and Stockner 1983; Murphy et al. 1986). In contrast, energy inputs from allochthonous sources decrease after harvest of riparian vegetation (Gregory et al. 1987; Bilby and Bisson 1992). Macroinvertebrate communities respond to these changes in food sources. Herbivorous invertebrates, particularly those that scrape algae from the substrate, are expected to become more abundant (but see Hawkins et al. 1982), while those species that feed on detritus (i.e., shredders, filterers, and collector-gatherers, sensu Merritt and Cummins 1984) typically decline in numbers (Hawkins et al. 1982; Beschta et al. 1995). The abundance of invertebrate predators has been shown to increase in response to increased secondary production in streams in the Oregon Cascades (Murphy et al. 1981; Hawkins et al. 1982). As riparian vegetation recovers, the amount of solar radiation reaching the channel diminishes, algal production decreases, and shredders and collector-gatherers begin to replace scrapers. It is important to note that these responses are likely typical only of streams where primary production is light-limited (i.e., small streams in dense forests); the response of invertebrates and juvenile salmonids to canopy removal in more open, eastside systems may differ.

The general pattern of change in aquatic communities in response to changing energy sources caused by logging of riparian vegetation can be confounded by other simultaneous changes in habitat conditions. Silt deposited from mass failures and surface erosion can affect invertebrate production as gravel interstices are filled by silt, and algae are buried or abraded (Beschta et al. 1995). In these instances, invertebrate assemblages are typically characterized by high numbers of a few tolerant, colonizing species (Newbold et al. 1980; Murphy et al. 1981; Hawkins et al. 1982; Lamberti et al. 1991). Loss of substrate complexity, including large woody debris, also tends to decrease the diversity of aquatic invertebrates. Similarly, application of insecticides and herbicides may have substantial and long-lasting effects on invertebrate community structure, with stoneflies, mayflies, and caddisflies all being particularly sensitive (reviewed in Beschta et al. 1995). Gregory et al. (1987) suggest that an overall pattern of increased production of a few taxa accompanied by a reduction in biodiversity may be common to all invertebrate trophic levels in streams that have been simplified through forest practices.

All of the logging-induced changes in physical and biological characteristics discussed in preceding sections interact to influence the composition and diversity of fish populations and communities; however, few studies in the Pacific Northwest have been designed to address specific relationships between changes in habitat attributes and structure of fish assemblage (Bisson et al. 1992). In addition, changes in one habitat attribute that benefit salmonid productivity (e.g., increased light and primary production) may be compensated by other adverse effects (e.g., loss of rearing habitat, sedimentation, excessive temperatures, change in timing of life-history events), which may not be manifest until subsequent life stages. Similarly, most studies of effects of logging in the Pacific Northwest have been conducted in relatively wet, forested ecosystems of the Coast Range and Western Cascades. In eastside systems, canopy removal, loss of physical structure, and increases in sediment loading may have greater ecological impacts on salmonids because different factors may limit production (e.g., summer temperatures, pool habitats).

Studies in the Cascades and Coast Ranges of Oregon and northern California indicated higher densities and biomass of salmonids in recently clear-cut reaches compared to shaded old-growth and second-growth reaches (Murphy et al. 1981; Hawkins et al. 1983); however, no analysis of age structure was provided. In western Washington, Bisson and Sedell (1984) found that total salmonid biomass was greater in streams that had been logged and cleaned (i.e., large woody debris removed), but that populations were dominated by underyearling trout, with proportionately fewer age I and older trout. These differences were attributed in part to a higher frequency of riffles (favored by underyearlings) and a lower frequency of pools (favored by older trout) in clear-cut reaches. Subsequent studies suggest that juvenile coho salmon and older trout, which prefer pool habitats, are adversely affected by conversion of pools to riffles (Bisson et al. 1992). In a long-term study of the effects of logging in an Oregon watershed (Alsea Watershed Study), numbers of fry migrating from a clear-cut section decreased by more than 50%, while prelogging and postlogging numbers from a patch-cut watershed and an unharvested watershed were not significantly different (Hall et al. 1987). Declines in the clear-cut watershed were attributed to a reduction in gravel quality from increased fines that led to a decrease in survival from egg deposition to emergence. In the same study, Moring and Lantz (1975) found reductions in late-summer densities of cutthroat trout in the clear-cut watershed following logging, but no reductions in the other two watersheds. Hartman and Scrivener (1990) reported that numbers of steelhead smolts declined, but cutthroat numbers remained constant following logging of 41% of the Carnation Creek watershed in British Columbia. In southeast Alaska, summer abundance of coho salmon increased following canopy removal; however, in winter parr densities

were lower in clear-cut reaches than old growth reaches, apparently because of a lack of large woody debris and undercut banks (Heifitz et al. 1986).

Other general effects of logging on fishes include decreased growth efficiency, reduced survival to emergence, increased susceptibility to disease, increased vulnerability to predation, lower winter survival, blockages to migration from poorly designed culverts, changes in the age-structure of fish populations, reduced development time of embryos, and altered timing of life-history events (Hicks et al. 1991a). Scrivener and Brownlee (1989) reported reductions in survival to emergence of coho and chum salmon of approximately 45% as a result of increased fine sediment in spawning gravels following logging of the Carnation Creek watershed. In the same watershed, Holtby et al. (1989) reported that increases in water temperatures following logging resulted in earlier outmigration of coho and chum salmon fry and earlier emigration by coho salmon smolts to the ocean. The change in migration timing of coho salmon smolts is believed to have reduced ocean survival. Furthermore, outmigrating coho smolts were evenly divided between 1- and 2-year old fish in years preceding logging, but dominated by 1-year old fish after logging. Such changes increase the vulnerability of specific year classes to environmental fluctuations in both the freshwater and marine environments. The Alsea Watershed study documented an increase in the number of early (November–January) coho migrants (Hicks et al. 1991a); whether the change was due to temperature-induced acceleration of growth or loss of rearing habitats remains uncertain. Nevertheless, these results suggest that small increases in temperature (1–2°C) can result in significant shifts in the timing of important life history events.

Finally, roads constructed for timber harvest may indirectly affect salmonids by increasing public access to previously remote locations. Angling pressure generally decreases with increasing distance from access roads; consequently, increased mortality from angling may accompany habitat degradation.

6.2 Grazing

Livestock grazing represents the second most dominant land use in the Pacific Northwest, following timber production. In Oregon, Washington, and Idaho combined, over 22.9 million hectares (56.5 million acres) of grassland and desert shrubland, approximately 3.2 million hectares (8 million acres) of nonfederal forest land, and an undetermined amount of Federal forest land are grazed by cattle and sheep (Pease 1985). This acreage represents approximately 41% of the total land base. Rangelands are fairly evenly divided between Federal and nonfederal lands; Federal rangelands total approximately 12.3 million hectares (30.5 million acres) (excluding Federal forest lands that are grazed), and nonfederal rangelands total 13.8 million hectares (34 million acres). Estimates from 1987 indicate that 4.76 million cattle and 0.87 million sheep were produced for sale in Idaho, Oregon, and Washington (Pease 1993). The majority of rangelands in Washington and Oregon lie east of the Cascade Range (Palmisano et al. 1993a), but livestock are also concentrated in the Willamette Valley and Puget Lowlands west of the Cascades, as well as in coastal valleys of Washington and Oregon. No estimates of rangeland area in northern California were available. However, from 1966–1980 California produced an average of approximately 4.75 million cattle and 1.0 million sheep annually (Hornbeck et al. 1983), a number comparable to the livestock production of Oregon, Washington, and Idaho combined. The largest concentrations of livestock in California within the current range of the Pacific salmon occur in the Sacramento-San Joaquin Valley, the Sierra Nevada and Coast Range foothill regions, and coastal valleys of northern California.

Livestock grazing in the West was already heavy by the mid-to-late 1800s. In 1898, the National Academy of Sciences prepared a report for the Interior Department alleging significant destruction by unregulated grazing in national Forest Reserves (Irwin et al. 1994). By the late 1920s, concern about deterioration of rangelands on national forests was growing (Platts 1991; Heady and Child 1994). In the 1930s the Forest Service documented widespread degradation of rangeland conditions, concluding that overgrazing had destroyed more than half of all rangelands and that 75% of remaining rangelands were degraded (Heady and Child 1994). Concern for rangeland conditions prompted Congress to enact the Taylor Grazing Act of 1934, which established 80 million acres of land in grazing districts to be administered by the U.S. Grazing Service, later to become the U. S. Bureau of Land Management (BLM) (Platts 1991; Wilkinson 1992). The percentage of total rangeland in "poor" condition decreased from 36% in 1936 to 18% in 1984, suggesting some improvement in overall range condition (Heady and Child 1994). However, recent reports have indicated that most riparian areas remain in fair-to-poor condition (Chaney et al. 1990; GAO 1991). Thus, while upland conditions appear to be improving, riparian areas continue to be degraded. In 1991, BLM began a program to improve riparian management, with a goal of restoring 75% or more of riparian areas to properly functioning condition by 1997 (Barrett et al. 1993).

Despite the generally poor condition of most riparian areas, the potential for restoring those areas damaged by grazing is arguably greater than for

those affected by other activities (Behnke 1977; Platts 1991). Recovery of grasses, as well as willows and other woody species, can occur within a few years when grazing pressure is reduced or eliminated (Elmore and Beschta 1987; Platts 1991; Elmore 1992). Restoration of fully functioning riparian areas that support a variety of plant species, including older forests of cottonwood and other large tree species, will take considerable time. Nevertheless, many important riparian functions—shading, bank stabilization, sediment and nutrient filtering, and allochthonous inputs—may be rapidly restored to the benefit of salmonids, provided the stress of grazing is alleviated and prior damage has not been too severe.

6.2.1 Effects on Vegetation

Heavy grazing around the turn of the century had significant and widespread effects, many of which persist today, on upland and riparian vegetation. Rangelands have experienced decreases in the percentage of ground covered by vegetation and associated organic litter (Heady and Child 1994). Species composition of plants in upland areas have shifted from perennial grasses toward non-native annual grasses and weedy species (Heady and Child 1994). East of the Cascade Range, upland sites that once supported plant associations of Idaho fescue are now devoid of native bunchgrasses, which have been replaced with tarweed, gumweed, and other noxious plants (Johnson et al. 1994). In riparian areas, willow, aspen, sedge, rush, and grass communities have been reduced or eliminated and replaced with annual grasses or sagebrush. Diaries of early trappers in eastern Oregon noted that grasses were as high as seven feet (Wilkinson 1992) and that streams were well lined with willows, aspen, and other woody vegetation (Elmore 1992). In eastside meadows, alteration of the vegetation has been so pervasive that little is known about the native vegetation that once inhabited riparian meadow communities. Currently, these meadows are dominated by Kentucky bluegrass, big sagebrush, and annual brome grasslands (Johnson et al. 1994). Fleischner (1994) recently reviewed the literature and found numerous examples of changes in species composition, diversity, and richness associated with livestock grazing or removal of livestock in western States (Table *6-5*).

Table 6-5. Deleterious effects of livestock grazing on plant communities in western North America. From Fleischner (1994). Reprinted with permission of Blackwell Scientific Publications, Inc.

Habitat	Location	Effect	Authority
Sonoran desert scrub	Arizona	Perennial grasses and Krameria (palatable shrub) showed dramatic density decreases with grazing	Blydenstein et al. (1957)
Mojave desert scrub	California	60% reduction in above-ground biomass of annuals, 16%–29% decrease' in cover of perennial shrubs with grazing	Webb and Stielstra (1979)
Sagebrush desert	Idaho	Grazed site had one-third species richness of ungrazed site	Reynolds and Trost (1980)
Desert grassland	New Mexico	Grass density increased by 110% after 30 years of protection from grazing	Gardner (1950)
Semidesert grassland	Arizona	Species richness increased as did canopy cover for midgrass, shortgrass, shrub, and forb groups after removal of livestock	Brady et al. (1989)
Semidesert grassland	Arizona	Woody plants significantly more abundant after removal of livestock	Bock et al. (1984)
Ponderosa pine forest	Washington	Decreased species richness on grazed sites	Rummell (1951)
Mountain canyon	Utah	Absence or near absence of 10 grass species on grazed sites	Cottam and Evans (1945)
Riparian	Oregon	Species richness increased from 17 to 45 species 9 years after removal of livestock	Winegar (1977)
Riparian	Arizona	Herbaceous cover of grazed plot less than half that of ungrazed plot	Szaro and Pase (1983)
Riparian	Colorado	Shrub canopy coverage increased 5.5 times, willow canopy coverage 8 times after removal of livestock	Schulz and Leininger (1990)

Much early alteration of rangelands was by settlers who engaged in widespread clearing of grasslands and riparian forests to grow crops, build houses, obtain fuelwood, and increase availability of land for domestic animals (Heady and Child 1994). Conversion of lands for the purpose of livestock production continues today. Woody shrubs and trees are sometimes removed by chaining and cabling—uprooting of vegetation with anchor chains or cables stretched between tractors—for the purpose of increasing grass production (Heady and Child 1994). Removal of woody shrubs through chemical application or by mechanical means is also a common practice in range management. In addition, suppression of fire on rangelands is responsible for changes in upland vegetation, including encroachment by juniper in many areas of eastern Oregon and Washington (Miller et al. 1989a).

Cattle and sheep affect vegetation primarily through browsing and trampling. Grazing animals are selective in what they eat; consequently, preferred vegetation types are generally removed first, followed by less palatable species. Heavy, continual grazing causes plants to be partially or wholly defoliated, which can reduce biomass, plant vigor, and seed production (Kauffman 1988; Heady and Child 1994). Selection of specific plant species may allow other taxa to dominate (Kauffman and Krueger 1984; Fleischner 1994). Vegetation may also be lost or damaged through trampling, which tears or bruises leaves and stems, and may break stems of woody plants. Regeneration of some woody vegetation, such as willow, cottonwood, and aspen, is inhibited by browsing on seedlings (Fleischner 1994). Vegetation may also be directly lost when buried by cattle dung. In a dairy pasture, MacDiarmid and Watkin (1971) found that 75% of grasses and legumes under manure piles were killed.

Livestock grazing also influences vegetation through modification of soil characteristics. Hooves compact soils that are damp or porous, which inhibits the germination of seeds and reduces root growth (Heady and Child 1994). Changes in infiltration capacity associated with trampling may lead to more rapid surface runoff, lowering moisture content of soil and hence the ability of plants to germinate or persist (Heady and Child 1994); however, in some instances, trampling may break up impervious surface soils, allowing for greater infiltration of water and helping to cover seeds (Savory 1988 in Heady and Child 1994). Soils in arid and semi-arid lands have a unique microbiotic surface layer or crust of symbiotic mosses, algae, and lichens that covers soils between and among plants. This "cryptogamic crust" plays important roles in hydrology and nutrient cycling (see Sections 6.2.3 and 6.2.6) and is believed to provide favorable conditions for the germination of vascular plants (Fleischner 1994). Trampling by livestock breaks up these fragile crusts, and reformation may take decades. Anderson et al. (1982) found recovery of cryptogamic crusts took up to 18 years in ungrazed exclosures in Utah. Finally, livestock indirectly affect plant species composition by aiding the dispersion and establishment of non-native species; seeds may be carried on the fur or in the dung of livestock (Fleischner 1994).

The effects of livestock grazing on vegetation are especially intense in the riparian zone because of the tendency for livestock to congregate in these areas. Gillen et al. (1984) found that 24%–47% of cattle in two pastures in north-central Oregon were observed in riparian meadows constituting only 3%–5% of the total land area. Roath and Krueger (1982) reported that riparian meadows that constituted only 1%–2% of the total land area accounted for 81% of the total herbaceous biomass removed by livestock. Similar preferences for riparian areas have been observed elsewhere in the west (reviewed in Kauffman and Krueger 1984; Fleischner 1994). Cattle and sheep typically select riparian areas because they offer water, shade, cooler temperatures, and an abundance of high quality food that typically remains green longer than in upland areas (Kauffman and Krueger 1984; Fleischner 1994; Heady and Child 1994). In mountainous terrain, the preference of cattle and sheep for the riparian zone also appears related to hillslope gradient (Gillen et al. 1984). Heady and Child (1994) suggest that cattle avoid slopes greater than 10%–20%.The intensity of use by livestock in riparian zones exacerbates all of the problems noted above and generates additional concerns. Alteration of flow regime, changes in the routing of water, and incision of stream channels can lead to reduced soil moisture in the floodplain. Many types of riparian vegetation are either obligate or facultative wetland species that are adapted to the anaerobic conditions of permanently or seasonally saturated soils. Stream downcutting and the concomitant lowering of the water table can lead to encroachment of water-intolerant species such as sagebrush and bunchgrasses into areas formerly dominated by willows, sedges, rushes, and grasses (Elmore 1992). In addition, flood events may be an important mechanism for seed dispersal throughout the floodplain for woody plants, a function that is diminished as channels are incised.

6.2.2 Effects on Soils

Rangeland soils are frequently compacted by livestock. The degree of soil compaction depends on soil characteristics, including texture, structure, porosity, and moisture content (Platts 1991; Heady and Child 1994). As a general rule, soils that are high in organic matter, porous, and composed of a wide range of particle sizes are more easily

compacted than other soils. Similarly, moist soils tend to be more susceptible to compaction than dry soils, although extremely wet soils may give way and then recover following trampling by livestock (Clayton and Kennedy 1985). The result of soil compaction is an increase in bulk density (specific gravity) in the top 5–15 cm of soil as pore space is reduced. Because of the loss of pore space, infiltration is reduced and surface runoff is increased, thereby increasing the potential for erosion (see Section 6.2.4). The available studies indicate that compaction generally increases with grazing intensity, but that site-specific soil and vegetative conditions are important in determining the response of soils to this grazing activity (reviewed in Kauffman and Krueger 1984; Heady and Child 1994).

Trampling by livestock may also displace or break up surface soils. In instances where surface soils have become impervious to water, light trampling may increase the soil's ability to absorb water. On the other hand, loosening soils makes them more susceptible to erosion. Heavily pulverized soil (dust) may become hydrophobic, reducing infiltration and increasing surface runoff. In arid and semi-arid climates, the cryptogamic crust has been shown to increase soil stability and water infiltration (Loope and Gifford 1972; Kleiner and Harper 1977; Rychert et al. 1978). Disruption of the cryptogamic crust may thus have long-lasting effects on erosional processes.

Livestock also alter surface soils indirectly by removing ground cover and mulch, which in turn affects the response of soils to rainfall. Kinetic energy from falling raindrops erodes soil particles (splash erosion), which may then settle in the soil interstices resulting in a relatively impervious surface. Livestock grazing can increase the percentage of exposed soil and break down organic litter, reducing its effectiveness in dissipating the energy of falling rain.

6.2.3 Effects on Hydrology

Grazing modifies two fundamental hydrologic processes, evapotranspiration and infiltration, that ultimately affect the total water yield from a watershed and the timing of runoff to streams. Loss of upland and riparian vegetation results in reduced interception and transpiration losses, thereby increasing the percentage of water available for surface runoff (Heady and Child 1994). Shifts in species composition from perennials to annuals may also reduce seasonal transpiration losses. Reductions in plant biomass and organic litter can increase the percentage of bare ground and can enhance splash erosion, which facilitates clogging of soil pores and decreases infiltration. Similarly, soil compaction reduces infiltration. Rauzi and Hanson (1966) report higher infiltration rates on lightly grazed plots, compared to moderately and heavily grazed plots in South Dakota. Similar experiments in northeastern Colorado showed reductions in infiltration in heavily grazed plots, but no differences between moderately and lightly grazed plots (Rauzi and Smith 1973). Johnson (1992) reviewed studies related to grazing and hydrologic processes and concluded that heavy grazing nearly always decreases infiltration, reduces vegetative biomass, and increases bare soil.

Decreased evapotranspiration and infiltration increases and hastens surface runoff, resulting in a more rapid hydrologic response of streams to rainfall. Some authors have suggested that the frequency of damaging floods has increased in response to grazing; however, there remains uncertainty about the role of grazing in mediating extreme flow events (reviewed in Fleischner 1994).

Reduced stability of streambanks associated with loss of riparian vegetation can lead to channel incision or "downcutting" during periods of high runoff. In naturally functioning systems, riparian vegetation stabilizes streambanks, slows the flow of water during high flow events, and allows waters to spread out over the floodplain and recharge subsurface aquifers (Elmore 1992). Moreover, riparian vegetation facilitates sediment deposition and bank building, increasing the capacity of the floodplain to store water, which is then slowly released as baseflow during the drier seasons (Elmore and Beschta 1987). Downcutting effectively separates the stream channel from the floodplain, allowing flood waters to be quickly routed out of the system and leading to lowering of the water table (Platts 1991; Elmore 1992; Armour et al. 1994). Consequently, summer streamflows may decrease even though total water yield increases in response to vegetation removal (Elmore and Beschta 1987). Li et al. (1994) found that streamflow in a heavily grazed eastern Oregon stream became intermittent during the summer, while a nearby, well vegetated reference stream in a similar-sized watershed had permanent flows. They suggested that the difference in flow regimes was a consequence of diminished interaction between the stream and floodplain with resultant lowering of the water table.

6.2.4 Effects on Sediment Transport

Livestock presence in the riparian zone increases sediment transport rates by increasing both surface erosion and mass wasting (Platts 1991: Marcus et al. 1990; Heady and Child 1994). Devegetation and exposure of soil by grazing facilitates detachment of soil particles during rainstorms, thereby increasing overland sediment transport. Rills and gullies often form in areas denuded by livestock trails or grazing,

resulting in increased channelized erosion (Kauffman et al. 1983). As gullies expand and deepen, streams downcut, the water table drops, and sediments are transported to depositional areas downstream (Elmore 1992; Fleischner 1994; Henjum et al. 1994). Stream downcutting leads to further desertification of the riparian area and promotes soil denudation and the establishment of xeric flora. This in turn increases the potential for soil erosion. Some evidence suggests that significant channel downcutting in the Southwest occurred prior to the introduction of livestock (Karlstrom and Karlstrom 1987 in Fleischner 1994); however, studies in eastern Oregon and northern California implicate livestock as a major cause of downcutting (Dietrich et al. 1993; Peacock 1994).

Mass wasting of sediment occurs along stream banks where livestock trample overhanging cut banks (Behnke and Zarn 1976; Platts and Raleigh 1984; Fleischner 1994). Grazing also removes vegetation that stabilizes streambanks (Platts 1991). Where banks are denuded, undercutting and sloughing occurs, increasing sediment loads, filling stream channels, changing pool-riffle ratios, and increasing channel width (Platts 1981 in Fleischner 1994).

6.2.5 Effects on Thermal Energy Transfer and Stream Temperature

Riparian vegetation shades streams and thereby regulates stream temperatures. On rangelands east of the Cascades, black cottonwood, mountain alder, and quaking aspen are the dominant deciduous tree species in natural communities, whereas west of the Cascades, black cottonwood, red alder, and big leaf maple are dominant (Kauffman 1988). Shrubby vegetation, such as willows, may also be important sources of shade along smaller streams and in mountainous areas (Henjum et al. 1994), and even tall grasses can provide some measure of shade along narrow first- and second-order streams (Platts 1991).

The removal of riparian vegetation along rangeland streams can result in increased solar radiation and thus increased summer temperatures. Li et al. (1994) noted that solar radiation reaching the channel of an unshaded stream in eastern Oregon was six times greater than that reaching an adjacent, well shaded stream and that summer temperatures were 4.5°C warmer in the unshaded tributary. Below the confluence of these two streams, reaches that were unshaded were significantly warmer than shaded reaches both upstream and downstream. A separate comparison of water temperatures at two sites of similar elevation in watersheds of comparable size found temperature differences of 11°C between shaded and unshaded streams (Li et al. 1994). Warming of streams from loss of riparian vegetation is likely widespread east of the Cascades and may be particularly acute because of low summer flows and a high percentage of cloud-free days.

The effects of riparian canopy in winter on stream temperatures are less well understood and various studies have shown increases, decreases, and no change in water temperature following removal of riparian canopy (reviewed in Beschta et al. 1987). Riparian cover can inhibit energy losses from evaporation, convection, and long-wave radiation during the winter, and several authors have suggested that removal of vegetation can increase radiative heat loss and facilitate the formation of anchor ice (Beschta et al. 1991; Platts 1991; Armour et al. 1994). This is most likely to occur in regions where skies are clear on winter nights and where snow-cover is inadequate to blanket and insulate streams (Beschta et al. 1987), primarily in mountainous regions and east of the Cascades and Sierra Nevada.

Alteration of stream temperature processes may also result from changes in channel morphology. Streams in areas that are improperly grazed tend to be wider and shallower than in ungrazed systems, exposing a larger surface area to incoming solar radiation (Bottom et al. 1985; Platts 1991). Wide, shallow streams heat more rapidly than narrow, deep streams (Brown 1980). Similarly, wide, shallow streams may cool more rapidly, increasing the likelihood of anchor ice formation. Reducing stream depth may expose the stream bottom to direct solar radiation, which may allow greater heating of the substrate and subsequent conductive transfer to the water.

6.2.6 Effects on Nufrienfs and Other Solutes

Livestock activities can directly affect nutrient dynamics through several mechanisms, The removal of riparian vegetation by grazing reduces the supply of nutrients provided by organic leaf litter. Livestock also redistribute materials across the landscape. Because riparian areas are favored by cattle and sheep, nutrients that have been ingested elsewhere on the range tend to be deposited in riparian zones or near other attractors, such as salt blocks (Heady and Child 1994). The deposition of nutrients in riparian areas increases the likelihood that elements such as nitrogen and phosphorous will enter the stream. Nutrients derived from livestock wastes may be more bioavailable than those bound in organic litter. Elimination of the cryptogamic crust by livestock may also alter nutrient cycling in arid and semi-arid systems. These microbiotic crusts perform the majority of nitrogen fixation in desert soils (Rychert et al. 1978). Loss of these crusts can reduce the availability of nitrogen for plant growth, potentially affecting plant biomass in uplands (Fleischner 1994).

Riparian areas play a major role in regulating the transportation and transformation of nutrients and other chemicals (see Section 3.9.6). As stream channels incise and streams are separated from their floodplains, soil moisture is reduced, which in turn alters the quantity and form of nutrients and their availability to aquatic communities. In the anaerobic environments of saturated soils, microbial activity transforms nitrate nitrogen (NO,) into gaseous nitrous oxide (N_2O) and elemental nitrogen (N_2) that are liberated to the atmosphere (Figure 6-2) (Green and Kauffman 1989). Under drier soil conditions (oxidizing environments), denitrification does not occur and nitrate-nitrogen concentrations in the soil increase. Because nitrate is negatively charged, it is readily transported by subsurface flow to the stream channel (Green and Kauffman 1989). Thus by altering the hydrologic conditions in the riparian zone, grazing can increase the amount of nitrate nitrogen released to streams. Excessive nitrate concentrations facilitate algal growth, increase turbidity, and in some cases cause oxygen depletion because of increased biochemical oxygen demand.

The form of other elements including manganese, iron, sulfur, and carbon also depends on the redox potential of soils. In their reduced form, manganese, iron, and sulfur can be toxic to plants at high concentrations (Green and Kauffman 1989). Obligate and facultative wetland plant species have special adaptations for coping with these reduced elements that allow them to survive where more xeric plants cannot. Thus, changes in hydrologic condition caused by downcutting can modify the form of elements available to plants, thereby altering competitive interactions among plants and changing riparian plant communities.

6.2.7 Effects of Vegetation Management

Fertilizers, herbicides, mechanical treatments, and prescribed fire are commonly used in rangeland management to alter vegetation in favor of desired species. In principle, the potential effects of these activities on salmonids and their habitats are no different than similar activities in forested environments; however, because the physical and biological processes that regulate the delivery of water, sediments, and chemicals to streams differ on forests and rangelands, so may the response of aquatic ecosystems.

Fertilizers are used on rangelands to increase forage production, improve nutritive quality of forage, and enhance seedling establishment, although the high costs and varied results have led to a decline in fertilizing rangeland in the past 20 years (Heady and Child 1994). Fertilizers that reach streams through direct application or runoff can adversely

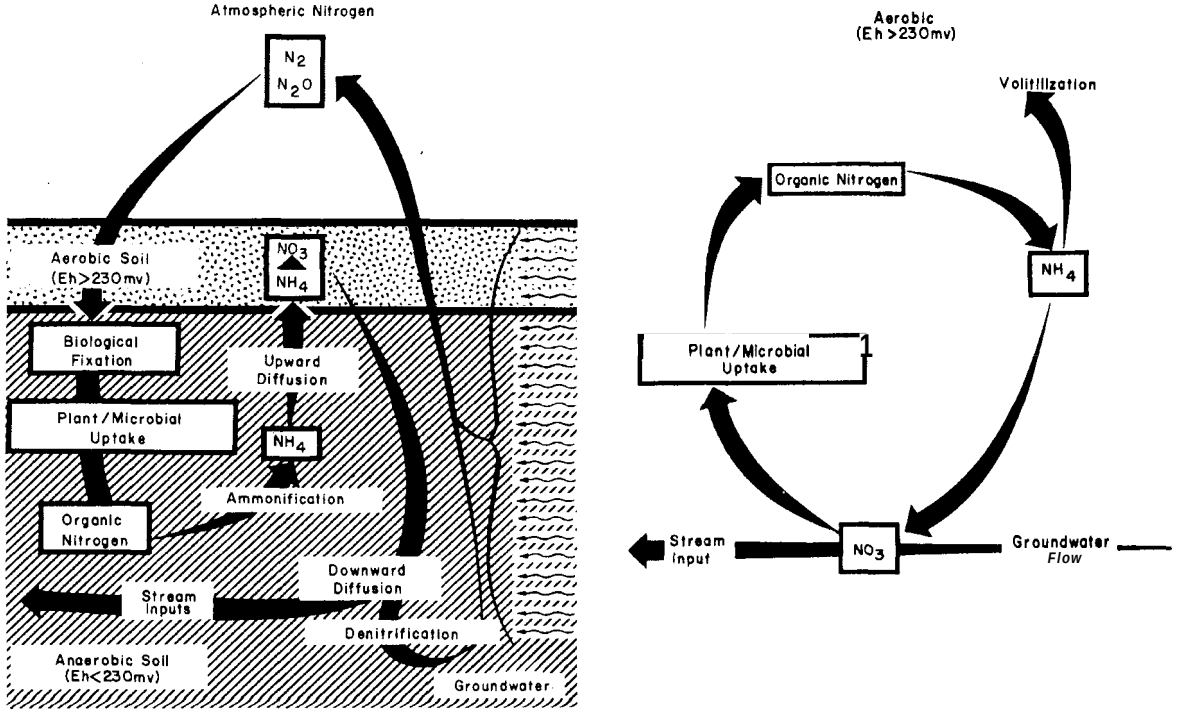


Figure 6-2. Nitrogen cycling pathways in undisturbed (left) and disturbed (right) riparian zones of northeastern Oregon, as indicated by redox potential (Eh). From Green and Kauffman (1989). Reproduced with permission from the principal author.

affect water quality. Nutrient enrichment (especially nitrogen) promotes algal growth, which in turn can lead to oxygen depletion as algae die and decompose. Conversely, fertilizer applied to rangelands may reduce sedimentation, hydrologic, and temperature effects by stimulating recovery of vegetation, including woody riparian shrubs. Herbicides are typically used to target unpalatable or noxious weeds that compete with desired forage species. Many of the herbicides commonly used in forestry (e.g., 2,4-D, picloram, glyphosate, tricopyr) are used in range management as well, although other highly selective herbicides may be used to control particular weeds common to rangelands, including unpalatable woody shrubs. Direct toxic effects on aquatic biota may occur where herbicides are applied directly to stream channels; however, risks of contamination can be minimized if adequate no-spray buffers are maintained (Heady and Child 1994). Herbicide applications to upland areas may decrease total groundcover, increasing the potential for surface erosion. In the riparian zone, use of herbicides may reduce production of deciduous trees and shrubs, opening streams to greater direct solar radiation, which in turn leads to elevated stream temperatures and increased algal production. These conditions can lead to insufficient nighttime dissolved oxygen concentrations and afternoon gas supersaturation. The loss of riparian vegetation also decreases the amount of organic litter and large wood delivered to streams. Furthermore, without the root structure of woody vegetation, banks are prone to collapse, increasing sedimentation and reducing the amount of cover for fish.

The influence of mechanical treatment and prescribed fire on aquatic ecosystems in rangelands depends on the type and intensity of disturbance. The use of tractors with dozer blades, brush rakes, cables, or rolling cutters for vegetation removal all can lead to compaction of rangeland soils (Heady and Child 1994) and thereby increase surface runoff and erosion. Disking of soils may break up relatively impervious soils, allowing greater infiltration of water; however, unless the area is rapidly revegetated, raindrop splash on exposed soils is likely to facilitate surface erosion and increase sediment delivery to streams. Disking and dozer use also rearrange soil layers, mixing topsoil with woody debris, which may affect re-establishment of vegetation. Positive effects of mechanical vegetation removal are also possible, Removal of vegetation with high evapotranspiration rates (e.g., juniper woodlands that have encroached because of grazing and lack of wildfire) may potentially increase the amount of water available during the summer, although documentation of this effect is poor. Prescribed fire is most likely to affect aquatic ecosystems through increased surface runoff and erosion resulting from the removal of vegetation and formation of hydrophobic soils.

In summary, manipulations of vegetation on rangelands can influence salmonid habitats through both direct and indirect pathways. These changes may harm or benefit salmonids depending on whether temperature, spawning sites, cover, or food limit the production of salmonids. Salmonid abundance will decrease if the increased invertebrate production is offset by undesirable alterations in the benthos assemblage to less nutritious species, reduced cover, increased sedimentation, and lower water quality.

6.2.8 Effects on Physical Habitat Structure

Livestock-induced changes in physical structure within streams result from the combined effects of modified hydrologic and sediment transport processes in uplands and the removal of vegetation within the riparian zone. Platts (1991) and Elmore (1992) review effects of grazing on channel morphology and are the sources of most information presented below. Loss of riparian vegetation from livestock grazing generally leads to stream channels that are wider and shallower than those in ungrazed or properly grazed streams (Hubert et al. 1985; Platts and Nelson 1985a, 1985b in Marcus et al. 1990). Loss of riparian root structure promotes greater instability of stream banks, which reduces the formation of undercut banks that provide important cover for salmonids (Henjum et al. 1994). Furthermore, the increased deposition of fine sediments from bank sloughing may clog substrate interstices, thereby reducing both invertebrate production and the quality of spawning gravels.. Over the long-term, reductions in instream wood diminish the retention of spawning gravels and decrease the frequency of pool habitats. In addition, the lack of structural complexity allows greater scouring of streambeds during high-flow events, which can reduce gravels available for spawning and facilitate channel downcutting. Figure 6-3 illustrates the characteristics of vegetation in functional and dysfunctional riparian zones on rangelands, and the channel modifications that typically result.

6.2.9 Effects on Stream Biota

As with forest practices, removal of riparian vegetation by livestock can fundamentally alter the primary source of energy in streams. Reductions in riparian canopy increase solar radiation and temperature, which in turn stimulates the production of periphyton (Lyford and Gregory 1975). In a study of seven stream reaches in eastern Oregon, Tait et al. (1994) reported that thick growths of filamentous algae encrusted with epiphytic diatoms were found in

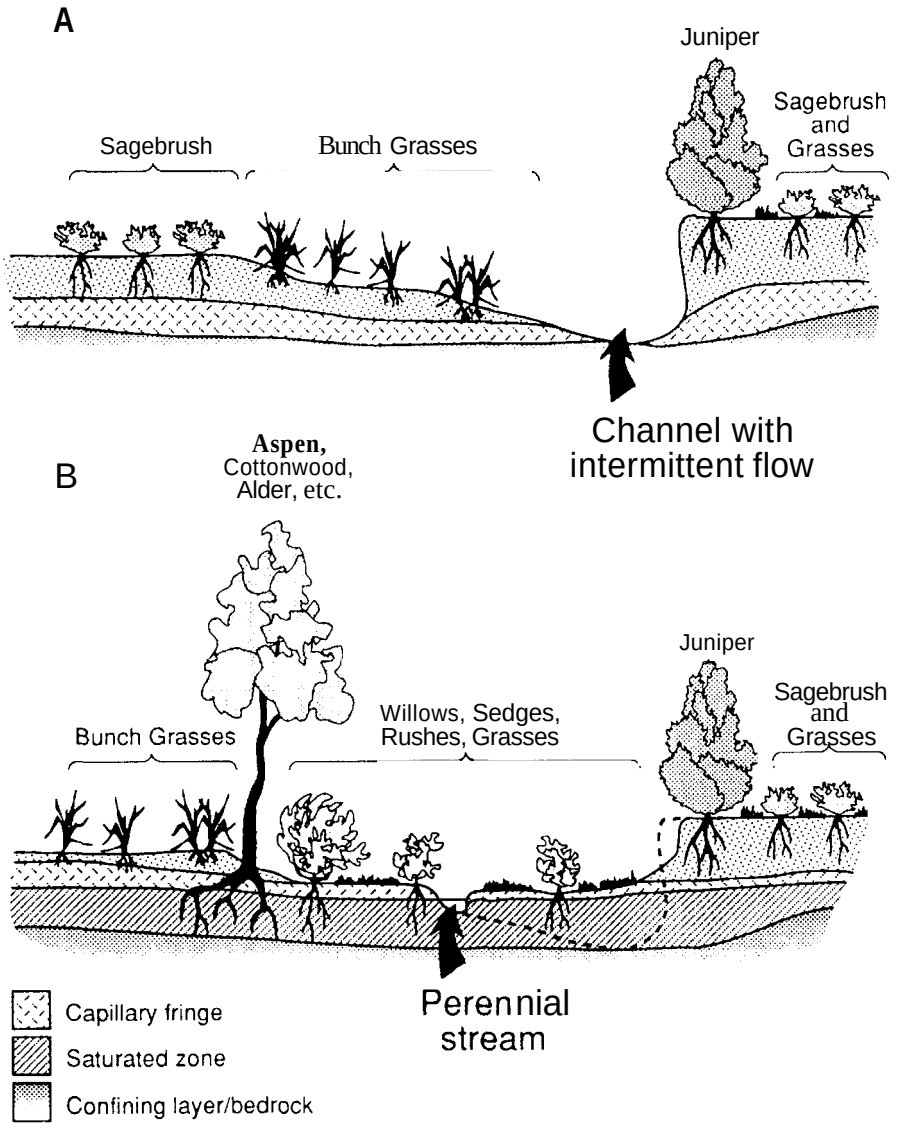


Figure 6-3. General characteristics and functions of a) disturbed and b) undisturbed riparian areas on rangelands. From Elmore (1992). Reproduced with permission from the publisher.

reaches with high incident solar radiation, whereas low amounts of epilithic diatoms and blue-green algae dominated in shaded reaches. Periphyton biomass was found to be significantly correlated with incident solar radiation.

While densities of macroinvertebrates in forested streams typically increase in response to increased periphyton production, the effect of stimulated algal growth in rangeland streams is less clear. Tait et al. (1994) found that biomass, but not density, of macroinvertebrates was greater in reaches with greater periphyton biomass. The higher biomass was a consequence of large numbers of *Dicosmoecus* larvae, a large-cased caddisfly that can exploit filamentous algae. Consequently, any potential benefits of increased invertebrate biomass to organisms at higher trophic levels, including salmonids, may be minimal, because these larvae are well protected from fish predation by their cases. Tait et al. (1994) suggest that these organisms may act as a trophic shunt that prevents energy from being transferred to higher trophic levels.

Evidence of negative effects of livestock grazing on salmonid populations is largely circumstantial, but is convincing nonetheless. Platts (1991) found that in 20 of 21 studies identified, stream and riparian

habitats were degraded by livestock grazing, and habitat improved when grazing was prohibited in the riparian zone. Fifteen of the 21 studies associated decreasing fish populations with grazing. Although they caution that some of these studies may be biased because of a lack of pregrazing data, the negative effects of grazing on salmonids seem well supported. Storch (1979) reported that in a reach of Camp Creek, Oregon, passing through grazed areas, game fish made up 77% of the population in an enclosure, but only 24% of the population outside the enclosure. Platts (1981) found fish density to be 10.9 times higher in ungrazed or lightly grazed meadows of Horton Creek, Idaho, compared to an adjacent heavily grazed reach. Within an enclosure along the Deschutes River, Oregon, the fish population shifted from predominately dace (*Rhinichthys* sp.) to rainbow trout over a ten-year period without grazing (Claire and Storch 1983). Platts (1991) cites other examples of improved habitat conditions resulting in increased salmonid populations.

6.3 Agriculture

Approximately 12% of the total land area in Washington, Oregon, and Idaho is dry cropland, with an additional 4% devoted to irrigated agriculture. Wheat, barley, and hay account for approximately 44% of the total harvested cropland, with fruits, nuts, berries, hops, peppermint, dry peas, and grass seed all contributing significantly to the total acreage (Jackson 1993). Like the other forms of food and fiber production, farming results in massive alterations of the landscape and the aquatic and riparian ecosystems contained therein. In general, the effects of agriculture on the land surface are more severe than logging or grazing because vegetation removal is permanent and disturbances to soil often occur several times per year. In addition, much agriculture takes place on the historical floodplains of river systems, where it has a direct impact on stream channels and riparian functions. Furthermore, irrigated agriculture frequently requires diversion of surface waters, which decreases water availability and quality for salmonids and other aquatic species (see Section 6.8). Qualitative summaries of the historical effects of agriculture on aquatic ecosystems have been reported by Smith (1971), Cross and Collins (1975), Gammon (1977), and Menzel et al. (1984).

6.3.1 Effects on Vegetation

In the Pacific Northwest, natural grasslands, woodlands, and wetlands have been eliminated to produce domestic crops. For example, in the Willamette Valley of Oregon, the original fire-maintained prairies and floodplain forests were replaced with croplands (Johannessen et al. 1971). Replacement of natural forest and shrubland vegetation with annual crops frequently results in large areas of tilled soil that become increasingly compacted by machinery and are only covered with vegetation for a portion of the year. Commonly, little or no riparian vegetation is retained along streams as farmers attempt to maximize acreage in production. While there is potential to restore agricultural lands to more natural communities, conversions to croplands in most instances have been permanent alterations of the landscape.

6.3.2 Effects on Soils

Agriculture involves repeated tillage, fertilization, pesticide application, and harvesting of the cropped acreage. The repeated mechanical mixing, aeration, and introduction of fertilizers or pesticides significantly alters physical soil characteristics and soil microorganisms. Further, tillage renders a relatively uniform characteristic to soils in the cropped areas. Although tillage aerates the upper soil, compaction of fine textured soils typically occurs just below the depth of tillage, altering the infiltration of water to deep aquifers. Other activities requiring farm machinery to traverse the cropped lands, and roads along crop margins, causes further compaction, reducing infiltration and increasing surface runoff. Where wetlands are drained for conversion to agriculture, organic materials typically decompose, significantly altering the character of the soil. In extreme cases, the loss of organic materials results in "deflation," the dramatic lowering of the soil surface. Soil erosion rates are generally greater from croplands than from other land uses (see Section 6.3.4), but vary with soil type and slope.

6.3.3 Effects on Hydrology

Changes in soils and vegetation on agricultural lands typically result in lower infiltration rates, which yield greater and more rapid runoff. For example, Auten (1933) suggested that forested land may absorb fifty times more water than agricultural areas. Loss of vegetation and soil compaction increase runoff, peak flows, and flooding during wet seasons (Hornbeck et al. 1970). Reduced infiltration and the rapid routing of water from croplands may also lower the water table, resulting in lower summer base flows, higher water temperatures, and fewer permanent streams. Typically, springs, seeps, and headwater streams dry up and disappear, especially when wetlands are ditched and drained.

Water that **is** removed from streams and spread on the land for irrigated agriculture reduces streamflows, lowers water tables, and leaves less water for fish. Often the water is returned considerable distances from where it was withdrawn, and the return flows typically raise salinity and

temperature in receiving streams. Extreme examples of this occur in many rivers east of the Cascades and in the Central Valley of California. The flows of these rivers are naturally low in late summer, but the additional losses from irrigation accentuate low flows. Reductions in summer base flows greatly degrade water quality because the water warms more than normal and causes increased evaporation, which concentrates dissolved chemicals and increases the respiration rates of aquatic life.

Streams are typically channelized in agriculture areas, primarily to reduce flood duration and to alter geometry of cropped lands to improve efficiency of farm machinery. Because peak flows pass through a channelized river system more quickly, downstream flood hazards are increased (Henegar and Harmon 1971). When channelization is accompanied by widespread devegetation, the severity of flooding is increased, such as occurred in the Mississippi Valley in 1993. On the other hand, channelization of streams leads to decreases in summer base flows because of reduced groundwater storage (Wyrick 1968), which can limit habitat availability for fish and increase crowding and competition. In more extreme cases, streams may dry completely during droughts (Gorman and Karr 1978; Griswold et al. 1978).

6.3.4 Effects on Sediment Transport

Because of the intensity of land use, agricultural lands contribute substantial quantities of sediment to streams. The Soil Conservation Service (SCS 1984) estimated that 92% of the total sediment yield in the Snake and Walla Walla River basins of southeastern Washington resulted from sheet and rill erosion from croplands—lands that accounted for only 43% of the total land area. The loss of vegetative cover increases soil erosion because raindrops are free to detach soil particles (splash erosion). Fine sediments mobilized by splash erosion fill soil interstices, which reduces infiltration, increases overland flow, and facilitates sheet and rill erosion. Agricultural practices typically smooth and loosen the land surface, enhancing the opportunity for surface erosion. When crop lands are left fallow between cropping seasons, excessive erosion can greatly increase sediment delivery to streams (SCS 1984). Mass failures are probably rare on most agricultural lands because slopes are generally gentle; however, sloughing of channel banks may occur in riparian zones in response to vegetation removal.

6.3.5 Effects on Thermal Energy Transfer and Stream Temperature

Removal of riparian forests and shrubs for agriculture reduces shading and increases wind speeds, which can greatly increase water temperatures in streams passing through agricultural lands. In addition, bare soils may retain greater heat energy than vegetated soils, thus increasing conductive transfer of heat to water that infiltrates the soil or flows overland into streams. In areas of irrigated agriculture, temperatures increases during the summer are exacerbated by heated return flows (Dauble 1994). These effects are discussed in greater detail on Section 6.1.5.

6.3.6 Effects on Nutrient and Solute Transport

Agricultural practices may substantially modify the water quality of streams. Omernik (1977), in a nationwide analysis of 928 catchments, demonstrated that streams draining agricultural areas had mean concentrations of total phosphorus and total nitrogen 900% greater than those in streams draining forested lands. Smart et al. (1985) found that water quality of Ozark streams was more strongly related to land use than to geology or soil. Exponential increases in chlorine, nitrogen, sodium, phosphorus, and chlorophyll-a occurred with increases in percent pasture in streams draining both forested and pastured catchments, and fundamental alterations in chemical habitat resulted as the dominant land use changed from forest to pasture to urban. Stimulation of algal growth by nutrient enrichment from agricultural runoff may affect other aspects of water quality. As algal blooms die off, oxygen consumption by microbial organisms is increased and can substantially lower total dissolved oxygen concentrations in surface waters (Waldichuk 1993). Nutrient enrichment from agricultural runoff has been found to significantly affect water quality in two rivers in interior British Columbia. Die-off of nutrient-induced algal blooms resulted in significant oxygen depletion (concentrations as low as 1.1 $mg \cdot L^{-1}$) in the Serpentine and Nicomekl rivers during the summer, which in turn caused substantial mortality of coho salmon.

6.3.7 Effects of Fertilizer and Pesticide Use

The two most commonly used agricultural chemicals, herbicides and nitrogen, are frequently found in groundwater in agricultural areas. McBride et al. (1988) report that atrazine is the herbicide most often detected in corn belt groundwater. In Oregon, groundwater nitrogen concentrations at or above health advisory levels were found in Clatsop, Marion, Deschutes, Morrow, Umatilla, Union, and Malheur counties, and elevated levels were reported for Multnomah, Linn, and Lane counties (Vomocil and Hart 1993). Because of the lack of a statistically representative sample of groundwater in the region's

agricultural areas, the degree and extent of contamination is unknown.

Unlike native vegetation, agricultural crops require substantial inputs of water, fertilizer, and biocides to thrive. Currently used pesticides, although not as persistent as previously-used chlorinated hydrocarbons, are still toxic to aquatic life. Where biocides are applied at recommended concentrations and rates, and where there is a sufficient riparian buffer, the toxic effects to aquatic life may be minimal. However, agricultural lands are also characterized by poorly-maintained dirt roads and ditches that, along with drains, route sediments, nutrients, and biocides directly into surface waters. Thus, roads, ditches, and drains have replaced headwater streams; but rather than filter and process pollutants, these constructed systems deliver them directly to surface waters (Larimore and Smith 1963).

6.3.8 Effects on Physical Habitat Structure

Agricultural practices typically include stream channelization, large woody debris removal, construction of revetments (bank armoring), and removal of natural riparian vegetation. Each of these activities reduces physical habitat complexity, decreases channel stability, and alters the food base of the stream (Karr and Schlosser 1978). Natural channels in easily eroded soils tend to be braided and meander, creating considerable channel complexity as well as accumulations of fallen trees. Large wood helps create large, deep, relatively permanent pools (Hickman 1975), and meander cutoffs; the absence of snags simplifies the channel. Channelization lowers the base level of tributaries, stimulating their erosion (Nunnally and Keller 1979). The channelized reach becomes wider and shallower, unless it is revetted, in which case bed scour occurs that leads to channel downcutting or armoring. Channel downcutting leads to a further cycle of tributary erosion. Richards and Host (1994) reported significant correlations between increased agriculture at the catchment scale and increased stream downcutting. Incised channels in an agricultural region were found to have less woody debris and more deep pools than nonincised channels (Shields et al. 1994).

6.3.9 Effects on Stream Biota

Agricultural practices also cause biological changes in aquatic ecosystems. In two States typified by extensive agricultural development and with extensive statewide ecological stream surveys, instream biological criteria were not met in 85% of the sites (Ohio EPA 1990; Maxted et al. 1994a). Nonpoint sources of nutrients and physical habitat degradation were identified as causes of much of the biological degradation. In another study, Maxted et al. (1994b) also demonstrated that the amount of shading had marked effects on stream temperatures and dissolved oxygen concentrations (e.g., Figure 6-4). In some agricultural stream reaches without riparian vegetation, the extremes exhibited in both temperature and DO would preclude the survival of all but the most tolerant organisms. Higher temperatures increase respiration rates of fish, increasing oxygen demand at the same time that oxygen is depleted by stimulated plant respiration at night. During daylight hours, high plant respiration (elevated by greater nutrient concentrations, higher temperatures, and lower flows) may produce gas supersaturation and cause fish tissue damage. Smith (1971) reported that 34% of native Illinois fish species were extirpated or decimated, chiefly by siltation, and lowering of water tables associated with drainage of lakes and wetlands. Although point sources were described by Karr et al. (1985) as having intensive impacts, nonpoint sources associated with agriculture were considered most responsible for declines or extirpations of 44% and 67% of the fish species from the Maumee and Illinois drainages, respectively. Sixty-three percent of California's native fishes are extinct or declining (Moyle and Williams 1990), with species in agricultural areas being particularly affected. Nationwide, Judy et al. (1984) reported that agriculture adversely affected 43% of all waters and was a major concern in 17%.

Modification of physical habitat structure has been linked with changes in aquatic biota in streams draining agricultural lands. Snags are critical for trapping terrestrial litter that is the primary food source for benthos in small streams (Cummins 1974), and as a substrate for algae and filter feeders in larger rivers. Benke et al. (1985) describe the importance of snags to benthos and fish in rivers with shifting (sand) substrates. Such systems, typical of agricultural lands, support the majority of game fish and their prey. Marzolf (1978) estimates 90% of macroinvertebrate biomass was attached to snags. Hickman (1975) found that snags were associated with 25% higher standing crops for all fish and 51% higher standing crops for catchable fish. Fish biomass was 4.8–9.4 times greater in a stream side with instream cover than in the side that had been cleared of all cover (Angermeier and Karr 1984). Gorman and Karr (1978) reported a correlation of 0.81 between fish species diversity and habitat diversity (substrate, depth, velocity). Shields et al. (1994) found that incised channels in agricultural regions supported smaller fishes and fewer fish species.

On a larger scale, habitat and reach diversity must be great enough to provide refugia for fishes during temperature extremes, droughts, and floods

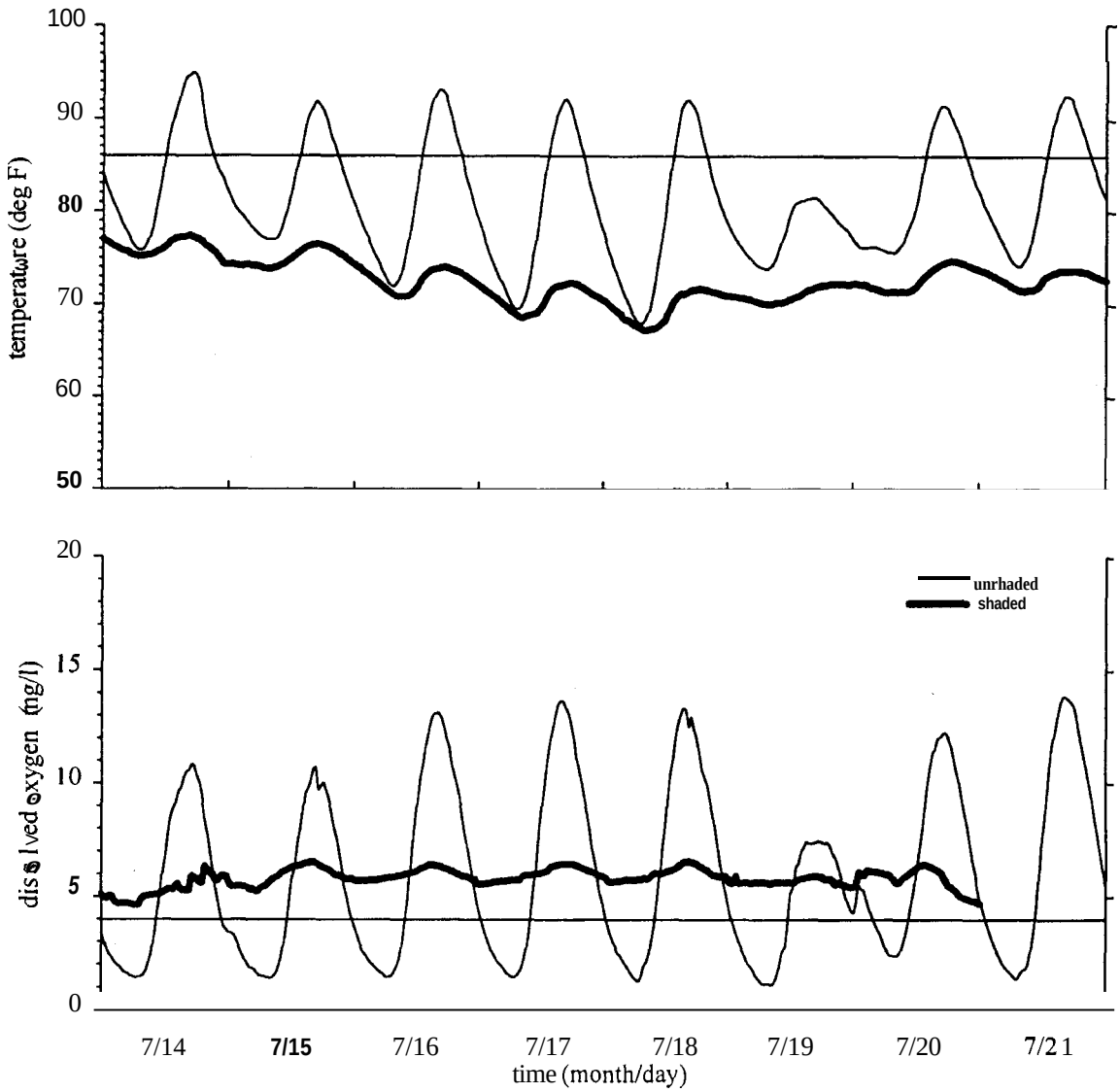


Figure 6-4. Diel fluctuations in temperature (top) and dissolved oxygen (bottom) in shaded and unshaded reaches of Mudstone Branch/Wharton Branch. From Maxted et al. (1994b). Reproduced with permission from the principal author.

(Matthews and Heins 1987). If refugia occur, fishes in agricultural streams can rapidly recolonize disturbed habitats and reaches. However, loss of refugia, alterations in water tables, simplifications of channels, and elimination of natural woody riparian vegetation symptomatic of agricultural regions creates increased instability and results in stream degradation (Karr et al. 1983).

6.4 Urbanization

Urban areas occupy only 2.1% of the Pacific Northwest regional land base (Pease 1993), but the impacts of urbanization on aquatic ecosystems are severe and long-lasting. Future projections suggest that urban areas will occupy an increasing fraction of the landscape. From 1982 to 1987, lands devoted to urban and transportation uses increased by 5.2% (45,346 hectares [123,813 acres]) in the Pacific Northwest. In the Puget Sound area, the population is predicted to increase by 20% between 1987 and the year 2000, requiring a 62% increase in land area developed for intense urbanization (PSWQA 1988). As urban areas continue to expand, natural watershed processes will be substantially altered.

Urbanization has obvious effects on soils and natural vegetation that, in turn, affect hydrologic and erosional processes, as well as physical characteristics of aquatic habitats. Urban

developments, including roads, buildings, sidewalks, and other impervious surfaces, greatly reduce water infiltration, which alters the routing and storage of water in the basin. Many of the resulting changes are intended and make the land more amenable to specific human uses (e.g., transportation, human habitation), but other important resource values (e.g., water supplies, fisheries, and wildlife) may be damaged by unintended effects on aquatic ecosystems, including increased peak flows, channel erosion, landslides, pollution, and channelization.

6.4.1 Effects on Vegetation

Urbanization causes severe and essentially permanent alteration of natural vegetation. The total vegetated area in the basin is typically diminished, and replacement vegetation (e.g., lawns, ornamental plants) often requires large quantities of water and fertilizers for growth. In addition, riparian corridors are frequently constricted, disabling or altering riparian processes. The loss of riparian vegetation reduces inputs of large woody debris and smaller organic detritus including leaves. Stream channels and banks are deprived of stability provided by large woody debris and the roots of riparian vegetation.

6.4.2 Effects on Soils

The effects of urbanization on soils can be divided into two phases. During urban construction, significant soil displacement, alteration, and movement occurs associated with grading, filling, and hauling activities. Once land conversion is complete, much of the surface soil is covered with buildings, concrete, and asphalt. In most residential areas, soils may be exposed, but they are generally altered and fertilized to support domesticated vegetation. Because of this dramatic alteration, the ecological functions that occur in the soil are likely greatly diminished, and these changes may be permanent.

6.4.3 Effects on Hydrology

Urbanization significantly influences hydrologic processes, increasing the magnitude and frequency of peak discharges and reducing summer base flows (Klein 1979; Booth 1991). These changes occur primarily because of increases in the impervious surface and the replacement of complex, natural drainage channels with a network of storm pipes and drainage ditches (Lucchetti and Fuerstenberg 1993). In urban areas, infiltration is reduced as 1) soils are stripped of vegetation, compacted, and or paved; 2) internal draining depressions are graded; 3) subsurface flow is intercepted by drains and discharged to streams; and 4) buildings are erected (Booth 1991). Instead of infiltrating into the soil, storm water is quickly delivered to the channel, resulting in a more episodic flow regime with higher peak flows and reduced base flows. In nonurban areas west of the Cascades, rainfall intensities are lower than the rate of infiltration, and subsurface flows predominate (Dunne et al. 1975). Only a small portion of the watershed contributes overland runoff; the remaining water infiltrates and becomes part of the subsurface regime. In arid and semi-arid eastside systems, overland runoff is more common because of higher rainfall intensities, sparse vegetation, and shallow, less permeable soils. Runoff generally travels quickly from the hillslopes to the channel, and virtually all parts of the watershed contribute to storm runoff. Although eastside runoff is primarily overland flow, urbanization increases the efficiency of water delivery to the channel. Culverts and drainpipes are straighter and provide a more direct and more efficient flow to the stream channel.

Increases in storm runoff caused by decreased infiltration also may result in more frequent flood events (Klein 1979). Using a model that incorporated historical storm data for Hylebos Creek, Washington, Booth (1991) found that over a 40-year simulation period, storm flows from an urban area were significantly greater than those from a forested basin. For the fully forested basin, seven floods exceeding the magnitude of a 5-year event were simulated for the 40-year period. In contrast, in the urbanized basin, simulated floods equaled or exceeded the discharge of a 5-year flood event in 39 of the 40 years (Figure 6-5).

Water withdrawals for water supply, industry and food processing can alter the flow regimes and quantity and quality of stream water. Muckleston (1993) reported that public water supplies accounted for 42% and 84%, respectively, of the total withdrawals from surface waters in the Willamette Basin, Oregon, and Puget Sound, Washington; these areas have the highest population densities found in these two States. In the lower Columbia sub-basin, public water supply and industrial usage make up over 80% of total withdrawals. East of the Cascade crest, food processing is generally the most significant industrial use of water though refining primary metals is important locally in the Clark Fork, Kootenai, Spokane, and mid-Columbia sub-basins. The need for water supplies, dependable power, and flood control has led to numerous impoundments on the major Northwest river systems. These reservoirs have altered the natural flow regimes and fish habitats. For example, flows in the Willamette River, which historically reflected annual precipitation patterns, have been substantially altered to accommodate urban water needs. On average, summer low flows are higher than in predevelopment periods because water is now stored during the wet season and released during the summer.

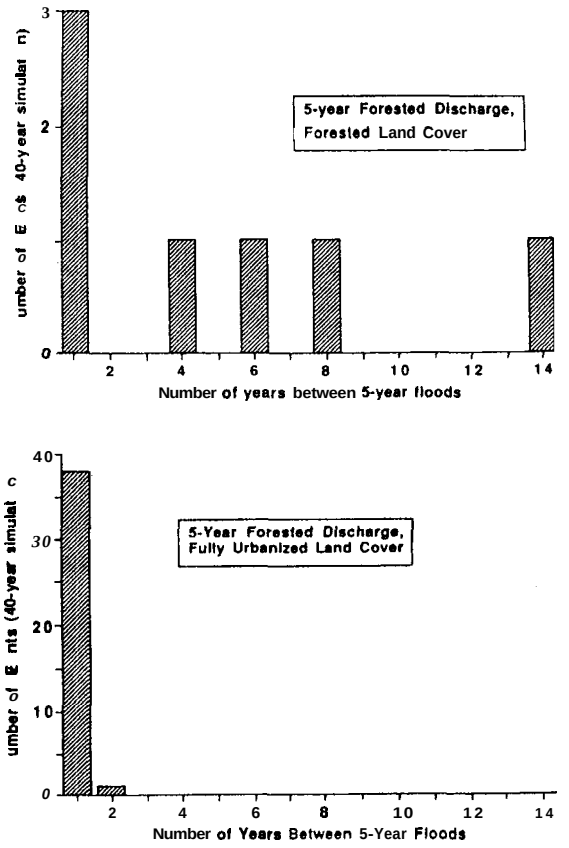


Figure 6-5. Hydrologic Simulation Program Fortran simulation of the Hylebos Creek basin in southwest King County, Washington, under fully forested land cover (top) and fully urbanized condition (bottom) (approximately 40% effective impervious area). Bars show the number of years separating discharge events of 5-year recurrence or greater. For forested condition, the average separation is 5 years (40 years of simulation, 8 events), but the actual spacing varies from 1 year (i.e., successive years) to 14 years. For urbanized condition, discharges exceeding 5-year forested event occur every year except one. Figure reprinted with permission from Booth (1991). ©1991 by the Institute for Environmental Studies, University of Washington.

6.4.4 Effects on Sediment Transport

Loss of vegetation and alteration of soil structure during construction of buildings and roads may increase sediment loading to streams by several orders of magnitude (Klein 1979); however, the effect is likely to be of short duration. Once building and landscaping is complete, surface erosion is reduced, possibly to levels lower than prior to construction because much of the land surface is under impervious surfaces. Specific effects are likely to vary with degree of urbanization, and whether drainage ditches are composed of erodible materials or concrete. Street sweeping and runoff from city streets transports some sediment to storm sewers and ultimately to streams, but the impact of that sediment is negligible. However, contaminants associated with such sediments can have significant impacts on water quality (see Section 6.4.7).

6.4.5 Effects on Thermal Energy Transfer and Stream Temperatures

Changes in riparian vegetation along urban streams can alter the degree of shading provided to the stream, which in turn influences seasonal and diurnal temperature ranges (see Section 3.7). As with other land uses, effects are likely to be greatest for smaller streams that previously had closed canopies. Published examples of changes in temperature regimes caused by urbanization are scarce; however, likely effects are increased maximum temperatures (Klein 1979), greater diel fluctuations, and reduced winter temperatures. Pluhowski (1970 in Klein 1979) found that winter stream temperatures in urban areas were 1.5–3°C lower than in nonurbanized streams on Long Island, New York. Although other land-use activities alter stream temperatures, in urban areas the loss of riparian function is long lasting.

Stream temperatures may also be indirectly affected by changes in hydrology, channel morphology, and the urban microclimate. Klein (1979) suggested that reductions in groundwater inflow may alter natural thermal regimes, resulting in lower winter minimum temperatures and higher summer maximum temperatures. Widening and shallowing of channels caused by greater peak discharges can also influence the rate of energy transfer to and from streams. Air temperatures in urban areas also tend to be warmer than those in surrounding rural areas, which may affect convective and evaporative energy exchange.

6.4.6 Effects on Nutrients and Other Solutes

The primary changes in nutrient cycling are the type and quantity of materials delivered to the stream channel. Large woody debris and leafy detritus are replaced in importance by nutrient loading from sewage and other sources. Novitzki (1973) reported that effluent from a sewage treatment plant in a small town in Wisconsin significantly degraded brook trout habitat downstream of the release point. High nutrient levels from the effluent generally stimulated primary and secondary production; however, under conditions of high temperature and low flow during the summer, heavy oxygen demand from the aquatic vegetation and effluent created critically low dissolved oxygen levels that resulted in fish kills. Omernik (1977) determined that total nitrogen exports from urban areas were second only to agriculturally influenced watersheds.

6.4.7 Effects of Chemical Use

Runoff from the urban areas contains many different types of pollutants depending on the source and nature of activities in the area. Wanielista (1978) identified numerous types of urban nonpoint source pollution including heavy metals, nutrients (phosphates and nitrates), pesticides, bacteria, organics (oil, grease) and dust/dirt. Heavy metal concentrations found in street runoff were 10–100 times greater than treated wastewater effluent, Contributions of grease and oil ranged from 32.8 $lb \cdot curb\ mile^{-1} \cdot day^{-1}$ for industrial areas to 4.9 $lb \cdot curb\ mile^{-1} \cdot day^{-1}$ for commercial areas (Pitt and Amy 1973). Residential areas fell between (18.6 $lb \cdot curb\ mile^{-1} \cdot day^{-1}$). Klein (1979) reported that 9% of persons that changed their own engine oil in their cars and disposed of used oil by pouring it into storm drains or gutters. In suburban areas, fertilizers, herbicides, pesticides and animal waste are added to the effluent. For example, Bryan (1972) found that pesticide loadings in runoff from urban areas was three-to-four times greater than for rural areas. In industrial areas, runoff may include heavy metals, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), high Ph concrete dust, and other toxic chemicals (Birch et al. 1992). Water quality degrades as a consequence of these pollutants entering streams, lakes, and estuaries. Biological oxygen demand is increased with the addition of organic materials, and lethal or sublethal effects may occur with influxes of heavy metals, pesticides, PCBs, and PAHs (see Sections 5.1.2 and 6.4.9).

6.4.8 Effects on Physical Habitat Structure

Urbanization frequently results in gross modification of stream and river channels through road construction, the filling of wetlands, encroachment on riparian areas and floodplains, relocation of channels, and construction and maintenance of ditches, dikes, and levees. Urban-related development can influence instream channel structure in a variety of ways. High densities of roads require road crossings, culverts, and other structures that constrain channels and may impede fish migration. Channels are frequently straightened in an attempt to route water quickly through the system and avert flood damage. Rip-rap, concrete, and other forms of channel revetment are commonly employed to counteract the increased erosive force associated with higher discharge volumes. In addition, with increased magnitude and frequency of floods in urban streams and rivers, greater within-stream bedload transport occurs, and channels become less stable (Bryan 1972; Scott et al. 1986). The rates of disturbance from flood events may accelerate to a point that the stream cannot recover between disturbance events. Lucchetti and Fuerstenberg (1992) noted that urbanized streams take on a clean "washed-out" look as channel complexity is lost. Such stream beds are uniform, with few pools or developed riffles, and have substrates dominated by coarser fractions rather than gravel, sand, and silt. The lack of large woody debris inputs exacerbates channel simplification (Lucchetti and Fuerstenberg 1992), causing increased bed scour and fill and changing channel hydraulics at a given maximum flow. These highly modified channels generally provide poor habitat for fish.

In unconstrained urban streams, stream channels may become substantially wider and shallower than streams in rural areas because of higher stream energy and increased erosion of streambanks (Klein 1979). In other areas, streambed morphology is further modified by channel incision, which leaves exposed, near-vertical channel banks (Lucchetti and Fuerstenberg 1992). In areas near the ocean, this can effectively isolate the estuaries from the surrounding riparian zone and essentially create a noninteracting

conduit between upriver areas and the sea, Important interactions between the stream and surrounding floodplain are lost.

6.4.9 Effects on Stream Biota

The structure of the biological community and the abundance of aquatic organisms are greatly altered by urban impacts on channel characteristics and water quality. Research indicates that stream quality impairment is correlated to the percentage of watershed imperviousness. Impaired water quality becomes noticeable at 8%-12% imperviousness and becomes severe above 30% imperviousness (Klein 1979; Pedersen and Perkins 1986; Limburg and Schmidt 1990). In a study of northern Virginia streams, Jones and Clark (1987) found that the taxonomic composition of macroinvertebrates was shifted markedly by urbanization, though development had minor effect on the total insect densities. Relative abundance of Diptera (primarily chironomids) increased at the most developed sites, and more sensitive orders, including Ephemeroptera (mayflies), Coleoptera (beetles), Megaloptera (dobsonflies), and Plecoptera (stoneflies), decreased. The response of Tricoptera (caddisflies) was variable. Pedersen and Perkins (1986) showed that a rural stream had twice the functional diversity of an urban steam. Those organisms that persisted were adapted to extreme bed instability.

Fish are also adversely affected by urbanization. Limburg and Schmidt (1990) demonstrated a measurable decrease in spawning success of anadromous species (primarily alewives) for Hudson River tributaries from streams with 15% or more of the watershed area in urban land use. In Kelsey Creek, Washington, urban development resulted in a restructuring of the fish assemblage in response to habitat degradation (Bryan 1972; Scott et al. 1986). Coho salmon appeared to be more sensitive than resident cutthroat trout to habitat alteration, increased nutrient loading, and degradation of the intragravel environment in the stream. In a study of Puget Sound streams, Lucchetti and Fuerstenberg (1992) found that fish assemblages in small urbanized streams have been dramatically altered or lost. They concluded that coho are of particular concern in urbanized areas because of their specific habitat needs (smaller streams, relatively low velocity niches, and especially large pools). Their study found that as impervious surfaces increased fish species diversity and coho abundance declined and resident cutthroat trout dominated.

Recent studies in the Pacific Northwest suggest that pollution from urban areas may be having insidious effects on anadromous salmonids. Arkoosh et al. (1991) found that juvenile chinook salmon that migrate through an urban estuary contaminated with PCBs and PAHs bioaccumulated these pollutants and exhibited a suppressed immune response compared to fish from an uncontaminated rural estuary. In subsequent studies, Arkoosh et al. (1994) exposed juvenile salmon collected from the same two estuaries, as well as their respective releasing hatcheries, to the pathogen *Vibrio anguillarum*. Salmon from the urban estuary exhibited higher mortality rate after 7 days than unexposed fish from the releasing hatchery. In contrast, no difference in mortality rates from this pathogen were observed between the salmon from the uncontaminated estuary and its releasing hatchery. Casillas et al. (1993) found that juvenile chinook exposed to PAHs and PCBs in an urban estuary showed suppressed immune competence and suppressed growth for up to 90 days after exposure, while juvenile chinook from a nonurban estuary did not develop these symptoms. They suggested that suppressed immune function, reduced survival, and impaired growth, result from increased chemical-contaminant exposure of juvenile chinook as they move through urban estuaries on their way to the ocean. The role of contaminants in the overall decline of salmonids is not known; however, these studies indicate that contaminant exposure is perhaps an overlooked cause of mortality for populations that migrate through urbanized streams, particularly because exposure occurs during the physiologically stressful period of smoltification.

6.5 Sand and Gravel Mining

Gravel and sand removal from streams and adjacent floodplains is common in many areas of the Pacific Northwest, particularly near and in low-gradient reaches of rivers west of the Cascade range. In Oregon, permits are required for removal of gravel or sand in excess of 38.3 m^3 (50 yr^3) (OWRRI 1995). Since 1967, the Oregon Division of State Lands has issued over 4,000 permits for gravel removal (OWRRI 1995), and between 1987 and 1989, a total of 1767 dredge, fill, and aggregate extraction permits were processed, 718 of which were new permits (Kaczynski and Palmisano et al. 1993a). Because there are no permit requirements for gravel extraction of less than 38.3 m^3, little information exists regarding the extent of small-scale gravel mining in Oregon. In Washington, large amounts of gravel are associated with glacial deposits and, thus, instream mining has decreased in recent years as extraction has shifted towards glacial and floodplain deposits (Dave Norman, Washington Department of Natural Resources, personal communication). Nevertheless, mining activity occurs near or in most major rivers west of the Cascade Range (Palmisano *et* al. 1993a, 1993b). Sandecki

(1989) reported that production of sand and gravel in California during 1986 exceeded 128 million short tons. The greatest demand for gravel and sand is associated with industrial development, and because of the expense of transporting gravel, mining is most prevalent around urban areas, along highways, or near other major construction sites. Most gravel permit sites in Washington are located near or in urban areas and along the Interstate 5 corridor (Figure 6-6). In Oregon, gravel production has generally risen between 1940 and 1990; however, gravel mining activity peaked during the 1960s and early 1970s with construction of the John Day, Green Peter, and Foster dams (OWRRI 1995). The majority of gravel mining in Oregon occurs in the Willamette Valley.

Two recent reviews focused on effects of gravel removal on hydrology and channel morphology (Sandecki 1989; Collins and Dunne 1990), and a third focused on effects on salmonids in Oregon (OWRRI 1995). Much of the material contained in this section comes from these three sources.

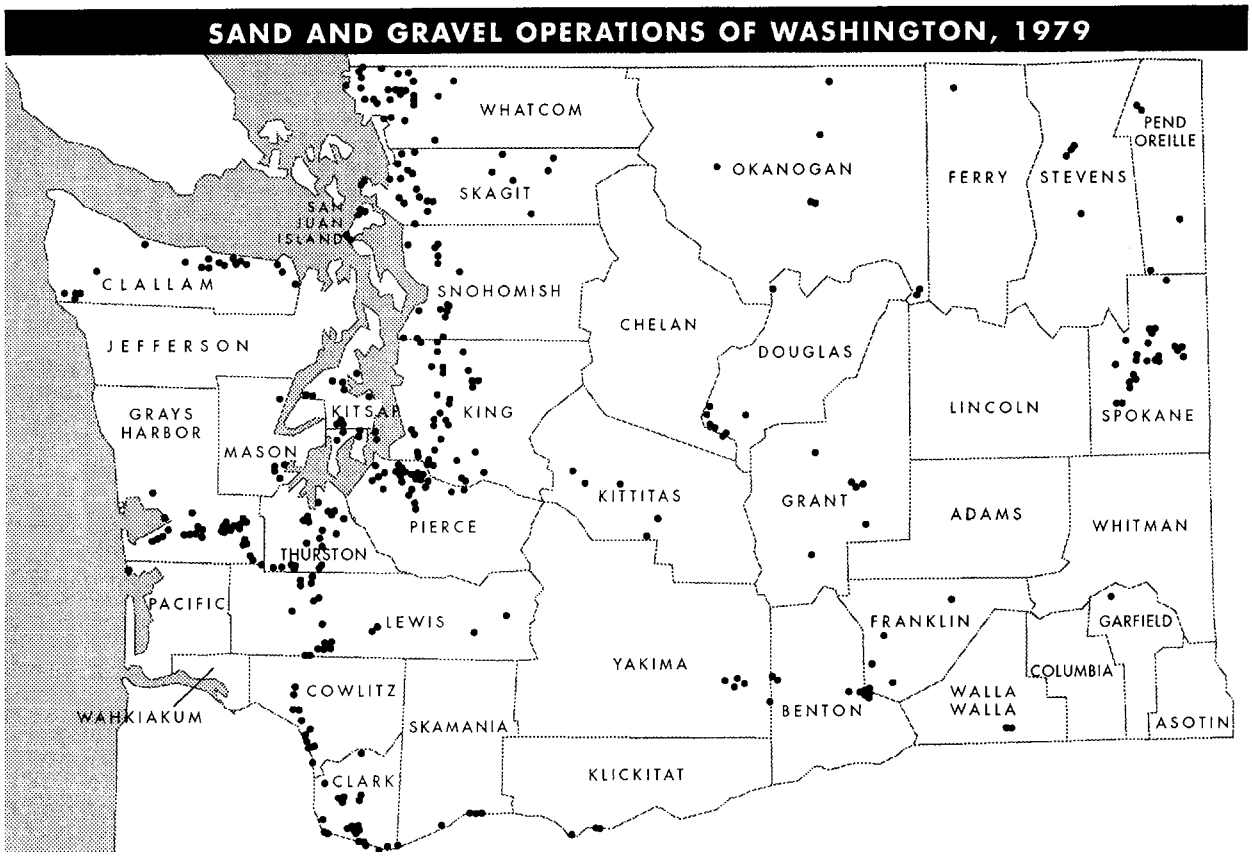


Figure 6-6. Sand and gravel operations of Washington, 1979. From Palmisano et al. (1993a). Reproduced with permission from the author.

6.5.1 Effects on Geomorphology and Sediment Transport

Removal of sand and gravel from within a stream channel may fundamentally alter the way in which water and sediment are carried through a system, resulting in altered channel morphology, decreased stability, accelerated erosion, and changes in the composition and structure of the substrate (Sandecki 1989; OWRRI 1995). The extent of effects depends on many site-specific characteristics, including the geomorphic setting (e.g., stream gradient and nature of bed material), the quantity of material extracted relative to the sediment supply, and the hydrologic and hydraulic conditions within the stream reach.

The effects of gravel mining on the stream environment involve a complex interplay between direct effects of channel modification and altered substrate composition, and the resulting alteration of erosional and depositional processes, which in turn feed back to cause further changes in channel configuration. Excavation of materials from the stream bed results in immediate changes to channel morphology. Newly created mining pits within streams are highly unstable and tend to migrate up or downstream in response to scouring and deposition of sediments (Lee et al. 1993). Thus, the physical effects of mining pits propagate away from the immediate excavation site (Sandecki 1989; OWRRI 1995). In undisturbed stream channels, coarser materials have a tendency, through hydraulic sorting, to "armor" the stream bed, increasing its resistance to scour (Lagasse et al. 1980). Finer materials work their way into deeper layers. Gravel mining disrupts the armor layer, leaving smaller materials at the bed surface that are more easily mobilized by streamflow; thus bedload movement occurs at lower stream velocities following gravel mining (Sandecki 1989).

Removal of bed material and increased bedload transport can combine to cause downcutting of the stream channel in both upstream and downstream directions (Sandecki 1989; OWRRI 1995). Downstream progression may result from reduced bed material discharge or decreased size of bed material, while upstream progression occurs when gravel extraction increases the river gradient (OWRRI 1995). In some cases, downcutting may occur until sand, gravel, and cobble are completely removed and underlying bedrock is exposed Downcutting may cause streambanks to collapse, introducing additional sediments into the stream (Collins and Dunne 1990).

Collins and Dunne (1990) recently reviewed case histories on the effects of gravel extraction on downcutting and found several examples where stream channels lowered 4 to **6** m in response to gravel mining (Table 6-6). In several cases, downcutting occurred over several decades; however, in one instance, a drop in bed elevation of **4.5** m occurred during two flood events that spilled into a large mining pit in Tujunga Wash, California, demonstrating that downcutting can occur rapidly under extreme circumstances. Kondolf and Swanson (1993) reported that gravel extraction below a dam in a Sacramento River tributary resulted in downcutting of more than 5 m and caused a shift from a highly braided channel to a single channel. Downcutting was severe in part because the dam prevented recruitment of gravels from upstream areas; however, reduced peak flows may have compensated for reduced sediment recruitment by reducing scouring. This example highlights the fact that effects of gravel mining depend on the cumulative effects of other activities in the watershed.

While the effects of off-channel mining are likely to be less direct, they may nevertheless be significant. Frequently, berms, dikes, or revetments are constructed to prevent flood flows from spilling into the excavation area and to reduce bank erosion. These structures prevent lateral migration of the stream channel, which may be important in recruiting gravels from streambanks. During high flows, water is constrained to a narrow channel, which increases the velocity and, hence, the erosive potential of the discharge. Artificially constricted channels, like excavations, may thus lead to degradation of the stream bed. Bar scalping also may affect erosion processes. When bar height is reduced, instream bars may be more prone to erosion when water level rises again (Collins and Dunne 1990).

In summary, the effects of gravel extraction on stream channels may include local adjustments, increased meandering or widening of the stream channel, changes in thalweg configuration, altered pool-riffle sequences, shifts from braided to single-thread channels, and downcutting of the channel bed (Sandecki 1989). Gravel mining may also change the frequency and extent of bedload movements and increase the amount of suspended fine sediments and turbidity in the water column. Turbidity caused by excavation generally decreases shortly after mining activity ceases; however, turbidity caused by changes in erosion potential may persist until the streambed restabilizes (reviewed in OWRRI 1995). Fine sediments may settle in gravel pits or travel downstream to settle in other slow-water areas. As a result, downstream substrates may be covered with sand, mud, and silt.

6.5.2 Effects on Hydrology

Gravel mining likely has little effect on the total amount of water moving through a stream system; however, it may significantly affect the routing and timing of streamflow. Both downcutting and channel simplification increase the hydraulic efficiency of the

stream—water is routed more quickly through the system, especially during periods of high flow (Sandecki 1989). This increased efficiency may reduce the probability of overbank flooding (Collins and Dunne 1990). The elimination of overbank flows prevents the recharge and subsequent release of water from the floodplain, which in turn results in flashier streamflows. In addition, channel downcutting may drain shallow groundwater, and lower the water table (Sandecki 1989; Collins and Dunne 1990; OWRRI 1995). Loss of shallow groundwater storage reduces summer base flows and may also lead to loss of riparian vegetation (Sandecki 1989).

Table 6-6. Case histories relating the effects of gravel extraction on channel morphology and hydrology of streams in Washington, Oregon, and California.

Location	Activitv	Effects
WASHINGTON		
Humptulips, Wynoochee, and Satsop Rivers*	Gravel bar scalping	Minimum rates of gravel extraction exceeded replenishment rate. Channel degradation (lowering) occurred at some sites.
White River'	Gravel extraction (partly for flood prevention), diking, and straightening	Aggradation in lower reaches, degradation in upper reaches.
Skykomish River†	In-channel gravel mining	Diminished size of gravel bars that were mined, as well as upstream and downstream sites. Reduced rate of bank erosion.
CALIFORNIA		
Cache Creek†	In-channel extraction during dry season	Channel degradation up to 9 m (avg 5 m) over 21-year period. Increased flood capacity has eliminated overbank flooding and is preventing soils from being deposited on flood plain. Drop in ground water table has shifted system from a "drain" system to recharge system. Loss of aquifer storage potential.
Russian River†	Gravel extraction	Channel degradation up to 6 m (avg 4 m). Exposure of bedrock substrate.
Dry Creek†	Gravel extraction	Channel degradation up to 4 m. Riparian vegetation has died, probably in response to lowering water table.
Tujunga Wash†‡	Off-channel gravel mining	Gravel pit was inundated by 1969 floods. Headward scour up to 4.5 m extended 790–914 m upchannel.
Redwood Creek†	Channelization, levee construction, gravel mining to low water level	Alternating lowering of bed by mining and raising of bed from redeposition. Shift in thalweg. Gravel bars removed annually by mining contributing to channel destabilization. Headward degradation of channel.
Stony Creek§	In-channel gravel mining	Channel shifted from braided configuration to single, incised, meandering channel. Degradation up to 5 m. Obliteration of natural low-flow channels. Effects modified by changes in flow regime due to construction of dam upstream.
OREGON		
Willamette River¶#	Sand and gravel extraction	Channel degradation of approximately 0.3 m per year over 20- to 30-year period. Degradation because of combined effects of sand and gravel extraction, bank stabilization, dams, watershed changes and natural geological events.

* Collins and Dunne (1989).
† Collins and Dunne (1990).
‡ Scott (1973).
§ Kondolf and Swanson (1993).
¶ Klingeman (1993).
\# OWRRI (1995).

6.5.3 Effects on Thermal Energy Transfer and Stream Temperature

The most likely changes in heat transfer processes resulting from gravel mining are increased heat exchange from the loss of riparian vegetation and alteration of the surface-to-volume ratio of the stream (OWRRI 1995). As discussed in Section 3.7, heat exchange is greater in wide, shallow streams than in narrow deep channels, so temperatures may increase or decrease depending on the specific change in channel morphology that follows gravel extraction. Stream temperatures may also increase because of inputs of heated water from off-channel ponds created by excavation (OWRRI 1995).

6.5.4 Effects on Nutrients and Other Solutes

We found no published information regarding the effects of instream gravel mining on nutrient cycling or availability. However, if the water table in the floodplain is lowered, floodplain soils may shift from reducing environments to oxidizing environments. Because the form of nitrogen and other solutes depends on the redox potential of the subsurface environment (Section 6.2.6) the availability of nitrate nitrogen and other solutes may increase in response to the oxidizing environment. Nutrient inputs are also affected where riparian vegetation is modified or eliminated.

6.5.5 Effects on Physical Habitat Structure

Most concern regarding the effects of gravel and sand mining on salmonids has focused on spawning habitats. Extraction of gravels may directly eliminate the amount of gravels available for spawning if the extraction rate exceeds the deposition rate of new gravels in the system. The areal extent of suitable spawning gravels may be reduced where degradation reduces gravel depth or exposes bedrock. In addition, decreases in the stability of streambeds can potentially increase embryo and alevin mortality because of scouring of gravel beds. Deposition of fine sediments downstream of mining activities may reduce the quality of these areas as spawning habitats (Kondolf 1994).

Widening and shallowing of stream channels in response to gravel mining may affect the suitability of stream reaches as rearing habitat for juveniles, particularly during summer low-flow periods, when deeper waters are important for survival. Similarly, a reduction in pool frequency may adversely affect migrating adults that require holding pools during their upstream migrations,

6.5.6 Effects on Stream Biota

Gravel mining can change the abundance and composition of species at lower trophic levels. Increased turbidity reduces light penetration, thereby affecting the production of benthic algae (OWRRI 1995). Aquatic invertebrates, which are an important prey for stream-dwelling salmonids, can be disrupted by disturbance of the substrate during mining (AFS 1988) or by changing substrate composition or covering of substrate with fine sediments (Hicks et al. 1991a). Potential effects on invertebrates include changes in species composition, reduced biomass, and slowed biotic colonization (OWRRI 1995).

OWRRI (1995) found few studies that address other effects of gravel mining on salmonids; however, qualitative inferences can be drawn from studies of the effects of logging, grazing, and other activities where physical changes are comparable. Salmonids require clean, well-oxygenated waters for successful incubation of embryos and alevins. Mechanical disturbance of spawning beds by mining equipment can potentially lead to high mortality rates of embryos and alevins. The OWRRI (1995) report cites one study where angler wading caused high mortality (43%–96%)of alevins with only one to two passes per day. It is likely that gravel mining equipment would be substantially more damaging to incubating embryos and alevins than anglers.

Turbidity reduces the reactive distance of fish during foraging (Barrett et al. 1992), clogs or damages buccal or gill membranes, and inhibits normal activities (Hicks et al. 1991a; Barrett et al. 1992). Sigler et al. (1984) reported that turbidities ranging from 25–50 NTU (nephelometric turbidity units) reduced growth and increased the tendency of young coho salmon and steelhead trout to emigrate from laboratory streams. Other direct effects of turbidity on fish are discussed in greater detail in Section 5.1.2. Potential effects on fish assemblages include reduced salmonid production, reductions in total biomass, decreased species diversity, and shifts away from species preferring clear waters towards species that are tolerant of high turbidities (OWRRI 1995). Those species that are most susceptible to increased fine sediments are those that rely heavily on benthic organisms for food or clean gravels for spawning, such as salmon and trout (OWRRI 1995).

6.6 Mineral Mining

In the Pacific Northwest and California, mining has had substantial influence on environmental conditions and patterns of human settlement. Mining provided the initial driving force for the ecological transformation of portions of the interior Northwest.

The discovery of gold in California and the western interior region in the 1860s catalyzed the large influx of people intent on extracting minerals from streams and mountainous slopes. This provided a wedge into the interior-montane ecosystems from the coastal regions for the cultural transformation of the Pacific northwest. Mining as practiced in the 1800s was especially disruptive to stream ecosystems, Hydraulic mining sluiced hillslopes down into streams, causing siltation of waterways and degradation of riparian habitats. Extensive cutting of inland forests was undertaken to provide trusses for mine tunnels and wooden viaducts, sluices, and flumes. By 1870, cattle and sheep that had been brought in to feed miners grazed throughout the intermontane Northwest (Robbins and Wolf 1994). Hydraulic mining of the main river valleys of California's Salmon River from 1870 to 1950 is estimated to have produced about 12.1 million m^3 (15.8 million yd^3) of sediments (PFMC Habitat Committee 1994). The effects there and elsewhere are still being felt today as sediments and pollutants derived from mine tailings continue to enter streams. The PSMFC (1994) reports that mining is responsible for polluting 19,350 km (12,000 miles) of rivers and streams in the western United States. Recovery rates of degraded streams vary, ranging from 20 years for areas with no acid drainage to generations for coal mines (with acid drainage), and radioactive phosphate and uranium mines (AFS 1988). Before the Surface Mining Control and Reclamation Act of 1977, little thought was given by many to post-mining effects and reclamation efforts (Nelson et al. 1991). However, some States (e.g., Oregon) have enacted more stringent laws regulating certain types of mineral extraction (Field 1993).

Minerals are extracted by several methods that can be combined into two broad categories. Surface mining includes dredging, hydraulic mining, strip mining, and pit mining. Underground mining utilizes tunnels or shafts to extract minerals by physical or chemical means. Surface mining probably has greater potential to affect aquatic ecosystems, although pollution associated with all forms of mining activities may be damaging to aquatic life. Specific effects on aquatic systems depend on the extraction and processing methods employed and the degree of disturbance.

6.6.1 Effects on Geomorphology and Sediment Transport

Like sand and gravel mining, mineral mining can have a significant effect on channel morphology, depending upon the extraction method. General effects of mining, including increased sedimentation, accelerated erosion, change in substrate, and increased streambed and streambank instability have been discussed in Section 6.5.1. Mineral mining can have some additional effects on channel formation and stability. During dredging operations, gravels are removed from rivers but are not hauled away from the channel; gold is extracted and waste gravels are piled along the banks, covering the riparian vegetation. These piles may eventually revegetate but remain unstable and leave banks with a high potential for erosion (Nelson et al. 1991). Dredging for gold in the early 1900s left extensive mine tailings, which continue to constrict stream channels and serve as chronic sediment sources (McIntosh et al. 1994b). Records from the 1940s indicate that substantial portions of the upper Grande Ronde river flowed beneath the extensive rubble left behind be early mining operations (McIntosh et al. 1994b). Hydraulic mining, which involves washing of unconsolidated ore-bearing alluvial gravels out of river banks or from hillslope areas down into the river, is uncommon today, yet effects are still being propagated throughout many river systems from long-abandoned operations (AFS 1988). Several forms of mining (strip, open-pit, quarry) remove the vegetation and topsoils from the site creating the potential for erosion and increased sedimentation. If topsoils are not retained to cover mine spoils, revegetation may be inhibited for extended time periods, especially if mine spoils are acidic (Butterfield and Tueller 1980; Fisher and Deutsch 1983).

6.6.2 Effects on Hydrology

Mineral mining may alter the timing and routing of surface and subsurface flows. Surface mining may increase streamflow and storm runoff (Sullivan 1967; Collier et al. 1970), as a result of compaction of mine spoils, reduction of vegetated cover, and the loss of organic topsoils, all of which reduce infiltration (Nelson et al. 1991). Merz and Finn (1951 in Nelson et al. 1991) reported infiltration rates of 452.1 $cm \cdot h^{-1}$ on undisturbed soils versus 43.2 $cm \cdot h^{-1}$ on adjacent graded spoils banks. Lower infiltration rates mean overland runoff and streamflow increase, particularly during storm events. Increasing flows may cause channel adjustments, including increased width and depth. Pit and strip mining may also affect groundwater by physical disruption of aquifers (Nelson et al. 1991). Large amounts of water are needed for processing mining products, and in arid regions east of the Cascades, withdrawals for mining may significantly affect the limited water supplies. Lindskov and Kimball (1984) estimated that extraction of 400,000 barrels of oil annually from oil shales in Utah, Colorado and Wyoming would require 86 million m^3 of water per year, which would be pumped from groundwater aquifers (Nelson et al. 1991). It was the

tremendous demand for water by mining operations that stimulated the water law of prior appropriation in the West (Wilkinson 1992).

6.6.3 Effects on Thermal Energy Transfer and Stream Temperature

Dredging and other mining practices may cause loss of riparian vegetation and changes in heat exchange, leading to higher summer temperatures and lower winter stream temperatures. Bank instability can also lead to altered width-to-depth ratios, which further influences temperature (see Section 3.7).

6.6.4 Effects on Nutrients and Pollutants

No published information was found regarding the effects of mineral mining on nutrient cycling or availability. However, surface mining and dredging likely affect inputs of nutrients where vegetation is removed or buried, and may increase nutrient spiraling length within streams where structure is simplified and nutrient retention is diminished.

Perhaps the most important effect of mining on aquatic ecosystems is contamination of surface waters from mine spoils. Acidification of surface waters by mining operations is generally considered to be the most serious consequence of mining. Water is acidified by oxidation of iron-containing waste products, which are then carried with runoff into local drainages (Nelson et al. 1991). In the western USA, much of the mineral recovery occurs from granitic deposits containing pyrite (Nelson et al. 1991). When exposed to atmospheric oxygen, pyrite is readily oxidized in water to produce sulfuric acid, which lowers the pH of mine spoils. Other metallic sulfides including chalcopyrite ($CuFeS_2$), sphalerite (ZnS), galena (PbS), and greenockite (CdS), undergo similar acid-generating processes (Nelson et al. 1991). Reductions in pH increase the mobility of many heavy metals (e.g., aluminum, arsenic, cadmium, chromium, cobalt, copper, iron, lead, mercury, nickel, zinc) by altering their chemical form, particularly if drainage is through waste piles. High acidity also facilitates formation of ferric hydroxide ($FeOH_3$), a noxious precipitate often called "yellow boy" (Nelson et al. 1991). The process of acidification is ongoing, and increased soil acidity converts metals into forms that are more bioavailable.

Nelson et al. (1991) reviewed the literature and found several examples of pollution associated with mine wastes. Levels of copper and zinc were 4–10 times above background levels 560 km from the major source of contamination on the Clark Fork River, Montana (Johns and Moore 1985). Duamie et al. (1985) reported that loadings from an abandoned mine in Montana were 13.6 $kg \cdot d^{-1}$ for copper and 1.6–145.5 $kg \cdot d^{-1}$ for zinc. Acid mine drainage and copper loadings of 41–147 $kg \cdot d^{-1}$ have been documented for Panther Creek, Idaho (Reiser 1986).

Heap leach mining is a form of open-pit mining used to extract gold from low-grade ore deposits. Extracted ore is crushed and placed into piles called pads where a dilute solution of sodium-cyanide ($NaCN$) is sprayed over the ore. As the cyanide solution percolates through the pad, gold is bonded to solutes and is collected in catch basins. With further processing, termed flotation, the gold is recovered. Cyanide is a well known toxicant, and any that leaches into local streams or is released from storage lagoons is potentially lethal to all aquatic organisms.

6.6.5 Effects on Physical Habitat Structure

The effects of surface mineral mining on the physical structure of salmonid habitats are similar to the effects of gravel and sand mining. Elevated levels of erosion increase sedimentation, which in turn affects the structure and composition of instream substrate. Spaulding and Ogden (1968) estimated that hydraulic mining for gold in the Boise River basin, Idaho produced 116,500 tonnes of silt in 18 months. They also reported that dredging in the Salmon River produced enough silt to cover 20.9 km of stream bottom with 0.16 cm of silt every 10 days, which reduced salmon spawning by 25%. Other effects of increased sedimentation include shallowing and widening of channels and reduction in pool frequency.

Dredging and placer mining practices have significantly altered the stability of habitats for fish and other organisms. An unnatural forced meander pattern was created along some sections of the Crooked River in Idaho, while another section was straightened. All along these disturbed sections, meadows and riparian vegetation were lost as a result of gold dredging (Nelson et al. 1991).

6.6.6 Effects on Stream Biota

Aquatic communities are affected by mining activities primarily through the alteration of physical processes (e.g., increased sediment inputs, greater channel instability, and simplification of channel structure) and chemical characteristics (e.g., acidification, heavy metals). Toxic effects of metals and acid can affect growth, reproduction, behavior, and migration of salmonids, resulting in the loss of sensitive species, changes in productivity, and alterations in population structure (AFS 1988). Increased turbidity reduces light penetration and decreases production of benthic algae (Nelson et al. 1991). Acidification of surface waters precipitates ferrous hydroxide, further decreasing benthic algal production and degrading macroinvertebrate habitat.

Stream acidification affects organisms that are sensitive to low pH, including salmonids. Salmonids exposed to low pH have been shown to experience reduced egg viability, fry survival, growth rate, development of pigmentation, ossification, and heart rate (Trojnar 1977; Nelson 1982). Johnson and Webster (1977) reported that spawning brook trout avoid areas of low pH, and speculated that recruitment is likely affected. Reduced numbers and diversity of benthic invertebrate taxa were found below an abandoned gold and silver mine on Coal Creek, Colorado (Reiser et al. 1982). Ephemeropterans (mayflies), plecopterans (stoneflies), and trichopterans (caddisflies) were found most sensitive to lowered pH in a study by Roback and Richardson (1969).

Lowered pH also enhances the availability and toxicity of heavy metals or metaloids. Arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, and zinc, are all toxic to fish, affecting growth, metabolism, respiration, reproduction, and numerous other biological functions (reviewed in Nelson et al. 1991). These substances may act singly, in combination, synergistically, or antagonistically (to reduce toxicity). Since many of these metals tend to bioaccumulate, increased toxicity is seen in higher-level trophic organisms for a given "background level" in surface waters. Numerous studies have developed LD„ levels for these toxicants using various invertebrate and fish (Table 6-7) test organisms. Other studies have also shown that continuous exposure to sublethal levels may produce effects that are just as important for determining ultimate species survival in the affected habitat (EPA 1986).

Effects of chronic pollution from mine wastes have been documented for several streams in the west. Mining wastes containing arsenic, cadmium, copper and zinc have been contaminating the Clark Fork River in Montana for more than 125 years. These metals have resulted in elevated metal concentrations in stream biota (Woodward et al. 1993) and are believed to be affecting benthic invertebrate communities and trout productivity in the river (Pascoe et al. 1993). Laboratory experiments in which rainbow trout fry were exposed to metal concentrations in water and food comparable to those in the Clark Fork indicate that uptake through the diet was the more important source of exposure (Woodward et al. 1993). Exposed fish experienced reduced growth and survival compared to control fish. Hughes (1985) found that Montana streams subjected *to* periodic mine effluents had fundamentally altered benthos assemblages and reductions or elimination of trout and sculpins. Other examples of exposure of salmonids and other aquatic organisms to pollution from mine wastes are reviewed in (Nelson et al. 1991).

6.7 Effects of Hydroelectric Dams

Hydroelectric dams have contributed substantially to the decline of salmonids in the Pacific Northwest, particularly anadromous stocks in the Columbia, Snake, and Sacramento River systems. The Northwest Power Planning Council (NPPC) estimates that current annual salmon and steelhead production in the Columbia River Basin is more than 10 million fish below historical levels, with **8** million of this annual loss attributable to hydropower development and operation (NPPC 1986). They conclude that approximately half of these losses occur during fish passage through the mainstem projects below Chief Joseph Dam (upper Columbia River) and Hells Canyon Dam (Snake River). The remaining 4 million in losses are due to the restriction of the fishes' range caused by dams; access to approximately 55% of the total basin area and 33% of the linear stream miles has been blocked by dams (Thompson 1976; PFMC 1979).

Dams influence salmonids and their habitats in a variety of ways. They impede migration of juvenile and adult fish, delaying migration (Raymond 1979) and thereby increasing the duration of exposure to predators. Juvenile or adult fish that pass through turbines may be killed outright or may be injured or disoriented, becoming easy prey for aquatic and terrestrial predators. Attempts to bypass dams by barging or trucking may stress fish and increase disease transmission among individuals (Bevan et al. 1994a), which ultimately may reduce survival.

Hydroelectric operations alter natural flow regimes, including daily and seasonal flow patterns. Unnaturally large daily fluctuations in flow occur downstream of dams during peaking operations. Seasonal flow fluctuations tend to be dampened, with water stored during periods of high flow in the winter or spring and released in summer when natural flows are lower (Marcus et al. 1990). These changes can affect migratory behavior of juvenile salmonids. Water-level fluctuations associated with hydropower peaking operations may reduce habitat availability, inhibit the establishment of aquatic macrophytes that provide cover for fish, and in some cases strand fish or allow desiccation of spawning redds (Palmisano et al. 1993a). The impoundment of water behind dams creates slackwater environments that are less favorable to salmonids. With the exception of the Hanford reach, virtually the entire lower and mid-Columbia River has been changed from a free-flowing river to a series of ponded reaches with little fast water, and significant spawning areas have been lost. The slow-moving water behind impoundments are also favorable to certain predators of salmonids, including northern squawfish (Faler et al. 1988).

Table 6-7. Reported toxicities of metals in soft water (< 45,000 $\mu g \cdot L^{-1}$ as $CaCO_3$). From Nelson et al. (1991). Reproduced with permission from the publisher.

Substance	Species	Reported Toxicity		Source
		Method*	Concentration ($\mu g \cdot L^{-1}$)	
Aluminum (Al)	Brook trout	LC50	3,600 – 4,000	Decker and Menendez (1974)
Arsenic (As)	Rainbow trout	LC50	10,800	Hale (1977)
Cadmium (Cd)	Rainbow trout	LC50	6.6	Hale (1977)
	Brook trout	MATC	1.7 – 3.4	Benoit et al. (1976)
Chromium (Cr)	Rainbow trout	LC50	24,100	Hale (1977)
	Rainbow trout	LC50	69,000	Benoit (1976)
	Rainbow trout	MATC	200 – 350	Benoit (1976)
	Brook trout	LC50	59,000	Benoit (1976)
	Brook trout	MATC	200 – 350	Benoit (1976)
Copper (Cu)	Coho salmon	LC50	46.0	Chapman and Stevens (1978)
	Rainbow trout	LC50	253.0	Hale (1977)
	Rainbow trout	LC50	125.0	Wilson (1972)
	Rainbow trout	LC50	57.0	Chapman and Stevens (1978)
	Rainbow trout	ILL	37.0	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	32.0	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	520	Sprague (1964)
	Brook trout	MATC	9.5 – 17.4	McKim and Benoit (1971)
Copper-zinc (Cu-Zn)	Atlantic salmon	TU	1,000	Sprague and Ramsay (1965)
Iron (Fe)	Brook trout	LC50	1,750	Decker and Menendez (1974)
Lead (Pb)	Rainbow trout	LC50	8,000	Hale (1977)
	Rainbow trout (eggs)	MATC	4.1 – 7.6	Davies et al. (1976)
Mercury (Hg)	Rainbow trout	LC50	33.000	Hale (1977)
Nickel (Ni)	Rainbow trout	LC50	35,500	Hale (1977)
Uranium (U)	Brook trout	LC50	2,800	Parkhurst et al. (1984)
Zinc (Zn)	Coho salmon	LC50	905.0	Chapman and Stevens (1978)
	Rainbow trout	LC50	1,775	Chapman and Stevens (1978)
	Rainbow trout	LC50	180 – 390	Finlayson and Ashuckian (1979)
	Rainbow trout	ILL	560	Sprague and Ramsay (1965)
	Atlantic salmon	ILL	92	Sprague (1964)
	Atlantic salmon	ILL	150 – 1,000	Zitko and Carson (1977)
	Atlantic salmon	ILL	420	Sprague and Ramsay (1965)
	Brook trout	MATC	534 – 1,360	Holcombe et al. (1979)

*LC50 = lethal concentration for 50% of test organisms; MATC = maximum acceptable toxic concentration; ILL = incipient lethal level: TU = toxic units.

Hydroelectric dams also modify sediment transport, natural temperature regimes, and the concentration of dissolved gases. Water storage at dams may prevent flushing flows that are needed to scour fine sediments from spawning substrate and move wood and other materials downstream. Behind dams, suspended sediments settle to the bottoms of reservoirs, covering coarser substrate and depriving downstream reaches of needed sediment inputs. The reduction in sediments downstream of dams leads to changes in channel morphology (Marcus et al. 1990). Reservoirs also modify temperature regimes in

streams and rivers. Below larger reservoirs that thermally stratify and that have hypolimnetic discharges, seasonal temperature fluctuations generally decrease; temperatures are cooler in the summer as cold hypolimnetic waters are discharged, but warmer in the fall as energy stored in the epilimnion during the summer is released. Finally, dams have resulted in changes in concentrations of dissolved oxygen and nitrogen concentrations (Bevan et al. 1994a). Behind dams, slow-moving water has lower dissolved oxygen levels than faster, turbulent waters. Water that spills over dams entrains air, and supersaturation of dissolved gases results. Gas supersaturation can cause gas bubble disease in salmonids, resulting in mortality, or weakening fish such that they become more vulnerable to predation or infection (Parametrix 1975; Blahm et al. 1975).

An exhaustive review of effects of dams on salmonids is beyond the scope of this document. A more thorough discussion of effects of dams on endangered salmonids in the Columbia Basin can be found in the recovery plan for Snake River salmon (Bevan et al. 1994a).

6.8 Effects of Irrigation Impoundments and Withdrawals

Damming and diversion of streams and rivers for agricultural purposes began in earnest in the mid-1800s as settlers moved into the region (Wilkinson 1992; Palmisano et al. 1993a). In the Pacific Northwest, withdrawals for agriculture (crop irrigation and stock watering) currently account for the vast majority (80%–100%)of offstream water uses in all major sub-basins east of the Cascades and in the upper Klamath Basin (Muckleston 1993). In addition, agriculture accounts for 62% of offstream water use in the coastal basins of Oregon, and 28% of the use in the Willamette Valley.

Water for irrigation is withdrawn in several ways. For major irrigation withdrawals, water is either stored in impoundments or diverted directly from the river channel at pumping facilities. Individual irrigators commonly construct smaller "push-up" dams from soil and rock within the stream channel, to divert water into irrigation ditches or to create small storage ponds from which water is pumped. In addition, pumps may be submerged directly into rivers and streams to withdraw water.

Many of the effects of irrigation withdrawals on aquatic systems are similar to those associated with hydroelectric power production, including impediments to migration, changes in sediment transport and storage, altered flow and temperature regimes, and water level fluctuations. In addition, aquatic organisms may be affected by pollutants from agricultural runoff and reduced assimilative capacity of streams and rivers from which substantial volumes of water are withdrawn. Alterations in physical and chemical attributes in turn affect many biological components of aquatic systems including vegetation within streams and along reservoir margins, as well as the composition, abundance, and distribution of macroinvertebrates and fishes.

6.8.1 Fish Passage

For many early irrigation dams, no fish passage facilities were constructed, resulting in the loss of several significant salmon runs. For example, irrigation dams in the Yakima River basin blocked sockeye runs estimated at 200,000 adult fish (Palmisano et al. 1993a). At some older irrigation impoundments (e.g., the Savage Rapids Dam on the Rogue River in Oregon), adult passage is hindered by poorly designed fish ladders (BR 1995). Smaller instream diversions may also impede the migrations of adult fish or cause juveniles to be diverted into irrigation ditches. Salmonid juveniles and smolts are also lost through entrainment at unscreened diversions or impingement on poorly designed screens.

6.8.2 Flow Modifications and Water-Level Fluctuations

The volume of water diverted for agriculture is substantial. Muckleston (1993) reports that withdrawals in the Snake River basin total approximately 45,000 acre-feet per day (equivalent to approximately 636.8 $m^3 \cdot s^{-1}$ (22,500 cfs); because this value is an annual average, daily diversions during the peak irrigation season are likely much higher. Diversion from individual rivers may also be great. For example, the Wapato Canal has a capacity to withdraw 57 $m^3 \cdot s^{-1}$ (2,000 cfs) from the Yakima River, with operation usually extending from March to mid-October (Neitzel et al. 1990).

Irrigation withdrawals affect both the total volume of water available to fish and the seasonal distribution of flow. Dams for irrigation typically store water during periods of high runoff in the winter or spring, and release water during the summer when flows are naturally low. Consequently, these impoundments tend to moderate streamflows, reducing winter and spring peak flows. Most direct diversions from rivers occur from spring to fall, during the peak growing season of agricultural crops. Because irrigation of crops coincides with periods of maximum solar radiation, evapotranspiration losses are greater than would occur under normal rainfall-runoff regimes, resulting in reduced summer flows in streams and rivers.

Changes in the quantity and timing of streamflow alters the velocity of streams which, in turn, affects

all types of aquatic biota. Water velocity is a major factor controlling the distribution of periphyton and benthic invertebrates in streams (Hynes 1970; Gore 1978; Horner 1978). At low velocities, diatom-dominated periphyton communities may be replaced by filamentous green algae (McIntire 1966). In western Washington streams, periphyton growth rates increased as velocity increased up to 0.1 $m \cdot s^{-1}$ (Gore 1978); however, as velocities increase above that level, erosion of periphyton exceeds growth. Reduced velocity may eliminate invertebrate species that require high velocities (Trotzky and Gregory 1974). The abundance and composition of fish species and assemblages is also regulated by the water velocity (Powell 1958; Fraser 1972). Changes in velocity influence incubation and development of eggs and larval fish by affecting oxygen concentrations within the gravel (Silver et al. 1963). Reduced water velocities in the Columbia River, which are in part a result of agricultural diversions, may delay downstream migration of salmon smolts. If temperatures become excessively warm, smolts may discontinue migration and revert to a presmolt physiology (Ebel 1977). Survival of these holdovers (fish delaying seaward migration for a year or more) is only about 20% (Adams et al. 1975), and very few may survive to return as adults (CRFC 1979).

Where irrigation water is withdrawn from smaller streams, seasonal or daily flow fluctuations may affect fish, macroinvertebrates in littoral areas, aquatic macrophytes, and periphyton (reviewed in Ploskey 1983). Lowered water levels may concentrate fish, which potentially increases predation and competition for food and space (Aggus 1979). Fluctuating water levels may delay spawning migrations, impact breeding condition, reduce salmon spawning area (Beiningen 1976), dewater redds and expose developing embryos, strand fry (CRFC 1979), and delay downstream migration of smolts. Water level fluctuations in reservoirs also reduce the density of bottom-dwelling organisms (Fillion 1967; Stober et al. 1976; Kaster and Jacobi 1978) through stranding, desiccation, or exposure to freezing temperatures (Powell 1958; Kroger 1973; Brusven and Prather 1974). In the littoral zone, frequent changes in water level can eliminate aquatic macrophytes that provide habitat for fish (Munro and Larkin 1950; Aas 1960). Loss of periphyton (attached algae) in the stream margins because of desiccation has been observed below hydroelectric dams (Neel 1966; Radford and Hartland-Rowe 1971; Kroger 1973) and may occur along the margins of streams below pumping facilities. Reductions in periphyton production affects other levels in the food web, particularly in large, unshaded rivers, where periphyton can be an important energy source.

6.8.3 Changes in Sediment Transport

Irrigation withdrawals and impoundments can affect the quantity of sediments delivered to streams and transported down river. In general, siltation and turbidity in streams both increase as a result of increased irrigation withdrawals because of high sediment loads in return waters. Unlined return canals contribute heavier silt loads than lined canals or subsurface drains (Sylvester and Seabloom 1962). Turbidity in the Wenatchee River doubled over a 45-year period because of increased agriculture and other human activities (Sylvester and Ruggles 1957). Once in the stream channel, the fate of sediments depends on hydrologic conditions. In systems where total water yield or peak discharge are reduced, sediments may accumulate in downstream reaches, affecting the quality of salmonid habitats. In the Trinity River in California, extreme streamflow depletion (85%–90% of average surface runoff) has allowed sediments to accumulate downstream, covering spawning gravels and filling in pools that chinook salmon use for rearing (Nelson et al. 1987). The lack of flushing flows during the winter has exacerbated this problem. In other systems, concentrations of suspended sediments below irrigation impoundments may be lower because slower water velocities allow sediments to settle (Sylvester and Ruggles 1957). The deposition of coarse, gravel sediments may be essential for developing high quality spawning gravels downstream of impoundments. Downstream reaches may become sediment starved, and substrate is frequently dominated by cobble and other large fractions unsuitable for spawning.

Iwamoto et al. (1978) reported that algae, phytoplankton, zooplankton, benthos, aquatic insects, and fish are all adversely affected by suspended and shifting sediments. In addition, sediments deposited into reservoirs, coupled with reduced streamflows, may improve habitat for intermediate hosts of several fish parasites. The impacts of suspended sediments, turbidity, and siltation are discussed in greater detail in Section 5.1.2.

6.8.4 Changes in Stream Temperature

Irrigation impoundments and withdrawals may increase water temperatures by increasing the surface area of rivers (i.e., reservoirs), reducing discharge volume, and returning heated irrigation waters to streams. In systems with irrigation impoundments, the seasonal thermal regime may also be altered. Reservoirs allow heating of surface waters that, depending on whether releases are from the epilimnion or hypolimnion, can result in increased or decreased temperatures. Below Lost Creek Dam on the Rogue River in Oregon—a multipurpose dam from which irrigation waters are withdrawn—

temperatures decreased during summer because of hypolimnetic discharges but increased during the autumn and winter as water that had been heated during the summer was released (Satterthwaite et al. 1992). The increases in fall and winter temperatures accelerated embryonic development of chinook salmon, resulting in earlier emergence. Typically, return flows of surface water from irrigation projects are substantially warmer after passage through the canals and laterals common to irrigated agriculture (Sylvester and Seabloom 1962). The degree to which water temperatures are affected by withdrawal of irrigation water ultimately depends on the proportion of water removed from and returned to the system and on the seasonal hydrologic regime. Water withdrawals in years of low flow are likely to have greater thermal effects on the fishes and other aquatic biota compared with similar withdrawals during years of high flow.

6.8.5 Changes in Dissolved Oxygen

Dissolved oxygen (DO) concentrations may decrease in both summer and winter in systems with irrigation withdrawals or impoundments. During summer, high solar radiation and warm air and ground temperatures combine to raise the water temperature of irrigation return flow, which diminishes the ability of water to hold DO. Increased water temperatures of irrigation return flows have been shown to reduce DO levels in the Yakima River (Sylvester and Seabloom 1962). Low summer flows can allow greater diel temperature fluctuations, which may exacerbate reductions in DO (McNeil 1968). In addition, higher concentrations of nutrients associated with irrigation returns may reduce DO by increasing biochemical oxygen demand. The extent and period of reduced DO concentrations depends on the quantity of water withdrawn and the quality of the return flow. In winter, low DO levels may occur in irrigation impoundments that have been drawn down. Fish kills can occur through anoxia if lowered water level facilitates freezing, which in turn inhibits light penetration and photosynthesis (Ploskey 1983; Guenther and Hubert 1993).

6.8.6 Influence of Impoundment and Water Withdrawal on Fish Diseases

Impoundment and water withdrawal for off-stream use may facilitate disease epizootics in salmonids by altering temperature regimes, lowering water levels, reducing flow velocities, creating habitat for intermediate hosts of parasites, and concentrating organisms, thereby facilitating the transmission of certain pathogens. Pathogen virulence and salmonid immune systems are greatly affected by water temperature (see Section 4.3.4); thus increasing temperatures by impoundment, flow reduction, or return of heated irrigation waters will affect disease susceptibility and prevalence in fish populations. Becker and Fujihara (1978) emphasize that extended periods of warm temperature and low flow increase the epizootiology of *F. columnaris* in Columbia River fish populations, and they warn that increasing withdrawal of Columbia River water for offstream use increases the potential for disease. Bell (1986) suggests that fish populations inhabiting lakes and reservoirs tend to experience more disease epizootics than fish species found in free-flowing rivers. Diseases in impoundments generally occur as a result of widespread parasite infections (Bell 1986). Decreasing water depth may provide additional habitat for intermediate hosts of parasites. Snail populations, as well as parasitic trematodes that use snails as intermediate hosts (e.g., *Diplostomum* and *Posthodiplostomum*), are more abundant in shallow waters (Hoffman and Bauer 1971). Consequently, reductions in flow may increase the likelihood of parasite epidemics. Finally, return flows from irrigated fields may transport parasitic nematodes and viruses from infested fields into streams (BR 1976).

6.9 River, Estuary, and Ocean Traffic (Commerciai and Recreationai)

Within a few decades of settlement, many estuaries and large low gradient rivers of western California, Oregon, and Washington were channelized. Eventually significant portions of major rivers, including the Columbia and Sacramento, were radically transformed.These systems were first altered for riverboat navigation and later to accommodate log rafting, barges, and ports (Maser and Sedell 1994). Navigation channels and pools continue to be maintained by dredging, removing snags, installing revetments, and operating locks and dams. Consequently, salmonids evolved in rivers and estuaries much different from what we now see in most of the Pacific Northwest.

What once was an incredible complex of channels, islands, bays, and wetlands connected with the sea are now highly simplified conduits. These complex mazes of shifting channels and bars laden with enormous snags and jams impeded navigation, but they were a haven for resident and migrating salmonids. Braided channels under gallery forests and flowing through alluvial plains with high water tables had abundant inflows of cool ground water during the summer. Water was stored in extensive floodplains (instead of behind dams) during the wet season and entered the channel via subsurface flows during the dry season. Network of essentially small, partially shaded rivers offered much more productive rearing habitat than the present navigation channels. These complex channels were nutrient-rich rearing habitats

in the summer and provided refugia from high streamflows during the winter. Changes in channel complexity are exemplified by the 80% reduction in the number of upper mainstem Willamette River channels documented by Sedell and Froggatt (1984).

High flows that once signaled migrations, offered passage over falls, and transported smolts rapidly to the estuaries have been moderated to facilitate the year-round boat traffic. For example, a 1938 Oregon statute directs that a year-round minimum of 6,000 cfs be maintained in the Willamette River at Salem (Muckleston 1993). As a consequence, numerous flood control reservoirs are operated to ensure this flow. The harbors, docks, and marinas offer some limited cover, but salmonids may be exposed to high levels of wood preservatives, petroleum, and organic wastes, as well as intense angling pressure. Noise pollution from boat traffic in estuaries and the open ocean may disrupt the navigation and communication of sharks and toothed whales; this may indirectly affect salmonids because sharks and whales feed on seals and sea lions, which in turn prey on salmon.

Channel changes have markedly altered the abundance and distribution of salmonids by making the physical habitat less suitable to resident and migrating fish. Losses of these salmon produced losses of particular life-history strategies from the population. Moreover, because these low gradient habitats were also among the most productive freshwater areas for salmon, their degradation has perhaps resulted in the loss of more fish than alterations of higher gradient reaches of similar size.

6.10 Wetland Loss/Removal

In 1989, Congress directed the Secretary of the Interior to assess the estimated total acreage of wetlands in each State in the 1780s and in the 1980s. The study (Dahl 1990) estimated that approximately 89.44 million hectares (221 million acres) of wetland functioned in the conterminous United States in the 1780s and that 53% of that area had been lost by the 1980s. Wetlands lost during this period included 1,839,741 hectares (4,546,000 acres) in California, 198,826 hectares (491,300 acres) in Idaho, 351,315 hectares (868,100 acres) in Oregon, and 166,734 hectares (412,000 acres) in Washington. These losses meant wetland area decreased from 4.9% to 0.4% of the land area in California, 1.6% to 0.7% of the land area in Idaho, 3.6% to 2.2% of the land area in Oregon, and 3.1% to 2.1% of the land area in Washington. These losses changed the function of ecosystems at the landscape scale because wetlands affect the transport and character of water in watersheds, lakes, and streams.

Wetlands provide a moderated climate compared to the adjacent uplands (cooler in summer and warmer in winter) because of the ground water (at relatively constant temperature) supplied to the site and the microclimate that develops within the vegetation occupying the wetland. Activities that modify the ground water supplied to the site, or modify the plant community, can impair the wetland's ability to moderate climate. Wetlands typically occur as a transition between upland and aquatic ecosystems, for example, at the edge of streams or between the stream and the adjacent valley walls. Wetlands require the surplus water that distinguishes them from uplands (EPA 1980b). Because wetlands may be only slightly wetter than adjacent upland, they are often targeted for drainage—either by ditching or tiling. These activities change the timing and duration of wetness of the site and modify or impair the wetland's functions. Diking may cause wetlands to be drier where the dike prevents floodwater from entering the wetland. Diking also may eliminate some functions performed by the wetland, for instance, floodwater storage; however, most wetlands do not exclusively depend on floodwater for their existence. Consequently, diking may not totally eliminate other normal wetland functions (as described below). Building, paving, or other permanent changes to the wetland's surface usually eliminate the majority of its functions, although some functions (e.g., floodwater storage) may continue at the site. Wetlands perform several functions related to hydrology, water quality, and habitat; these functions ultimately support salmonids.

6.10.1 Wetlands and Hydrology

Wetlands store water during runoff events, thereby reducing flood volumes and flood stages downstream. Further, floodwaters slow as they move into wetlands, reducing damage associated with scour and erosion caused by high velocity flows and allowing sediments, particulate organic matter, and other materials to be deposited in the wetland. Water quality improves with deposition of sediments, and some dissolved materials are either trapped within sediment deposits or utilized by vegetation and organisms in the wetlands. Movement of water through the wetland may also redistribute organic and inorganic particulates as well as import or export plant propagules or organisms. Infiltration of the flood waters into wetland soils supports other wetland functions, such as nutrient cycling, the retention and processing of elements and compounds, and the support of microbial communities adapted to survival in anaerobic conditions. And finally, because of unique hydrologic characteristics and soils, wetlands support unique floral and faunal communities. Wetlands are an integral component in the hydrologic cycle locally and of the habitat provided by the total watershed.

Novitzki (1979) reported that wetlands had a pronounced influence on flood peaks and seasonal distribution of streamflow. In Wisconsin, flood flows were 80% lower in watersheds with 40% lake-and-wetland area than in watersheds with no lake-or-wetland area. (Wetlands occupied 14.8% of the land area of Wisconsin in the 1980s [Dahl 1990], so wetlands may have a greater influence on streamflow in Wisconsin than in the Pacific Northwest.)

Wetlands also modify the rate of ground-water discharge to streams (Novitzki et al. 1993). Wetlands, particularly those occurring adjacent to streams, usually exist because of ground-water discharge. Wetland soils typically are less permeable than upland soils, especially where prolonged wetness fosters the accumulation of organic material (Novitzki 1989). Because the wetland soils are less permeable, the rate of ground-water discharge from upgradient sources, through the wetland, and to the stream is slowed. The net effect is to reduce the rate of ground-water discharge to the stream but to increase the length of time that discharge occurs. Ground water typically discharges at a relatively constant, cool temperature, and it has a major influence on the temperature regime in streams, especially low-flow periods in summer. Changing the amount and timing of ground-water discharge may change the temperature regime of the stream significantly, affecting the suitability of the stream as salmon habitat. Ground-water upwelling into streams through gravels is a determinant in selection of spawning redd for some species. The constant upwelling of fresh, oxygenated water may be necessary for egg and fry survival. Loss of wetlands will likely change the rate of ground-water discharge at critical times and may reduce spawning success in streams.

6.10.2 Wetlands and Water Quality

Wetlands retain particulate materials transported into them by overland flow or river flooding. Wetlands typically are flat areas adjacent to streams, and as floodwaters enter, flow velocities decrease and sediment loads are deposited. This phenomenon manifests as berms, often wooded, that build up next to the river channel in wide river valleys. Wetlands tend to stabilize stream banks because of the robust plant community that grows there. Wetlands tend to be wet through a larger part of the growing season, fostering plant growth that in turn provides sufficient root mass to stabilize soils. Where banks are stabilized by the lush wetland vegetation, stream channels tend to be somewhat deeper, and undercutting provides shelter to salmonids and other aquatic biota. Logging, grazing, farming, or other activities that change the wetland plant community can significantly reduce the wetland's ability to stabilize stream banks. Moreover, the velocity of water moving through wetlands is further reduced by dense vegetation, especially shrubs and trees, which in turn increases sediment deposition in the wetland. Thus, wetlands tend to reduce the amount of sediment transported to streams. Loss or removal of wetland areas may result in increased sediment loads (especially clays and silts) in receiving streams.

Wetlands also retain and process dissolved materials contained in overland flow or floodwaters. Some nutrients, as well as toxic substances, are taken up by plants, while others are bound to suspended solids, which subsequently settle to the bottom. Thus, loss or removal of wetland areas may result in increased nutrient and contaminant loading to receiving streams.

6.10.3 Wetlands and Salmonid Habitat

Wetlands may contribute significantly to certain characteristics required by salmonids in their aquatic ecosystems, such as variable, but moderate streamflows; cool, well oxygenated, unpolluted water; relatively sediment-free streambed gravel; an adequate food supply; and instream structural diversity provided by woody debris (Cederholm 1994). Because wetlands affect flood flows and springtime flows, they also influence the streamflow characteristics of the streams and aquatic habitat that support salmonids. Loss of wetlands likely increases the amount of individual flood peaks but reduces the duration of high-flow events. Streams in the Pacific Northwest may require the infrequent (i.e., the 100-year) flood to reset; however, they may also require stability between extreme events to recover and re-establish equilibrium. Wetland loss may reduce the time between significant (e.g., 5- to 50-year frequency) floods and impair the stream's ability to recover. For some salmonids, the timing and amount of streamflow triggers the movement of adult salmon into spawning streams, as well as the movement of fry and smolts downstream. Changing the timing of lows may thus subtly change the timing of migration and spawning, resulting in disruption of natural biological cycles. Changing the timing of spawning may result in minor, but *significant*, changes in the size and condition of salmon smolts returning to the ocean. These changes may have pronounced impacts on survival of young salmon in the ocean phase of their life cycle.

Wetlands support unique floral and faunal communities. The unique biota supported in wetlands contribute to the food web supporting the salmonids and associated biota in the streams, both adjacent to and downstream of the wetlands. Riparian vegetation, including that in wetlands, regulates the exchange of nutrients and material from upland forests to streams and wetlands (Cederholm 1994). Wetlands and ponds

have been found to provide critical habitats for both juvenile salmonids (Peterson 1982; Cederholm and Scarlett 1982) and a variety of wildlife species (Zarnowitz and Raedeke 1984). Species that frequent riparian areas include amphibians, reptiles, birds, mammals, and mollusks (FEMAT 1993). Activities that prevent the normal wetland functions or impair the connectivity of the wetland to the aquatic ecosystem may prevent the transport of materials into and out of the wetland, altering important elements of aquatic ecosystems. Interrupting or otherwise changing the connections between the wetland and the stream can impede the exchange of nutrients, organic detritus, insects, or other materials supporting the food web of the aquatic ecosystem.

Wetlands often provide refugia within the landscape. Especially in urban areas, agricultural areas, or other disturbed environments, wetlands are least suitable for conversion to other use: they often are left intact until all other lands have been converted. In highly modified landscapes, they may be the only natural areas left to provide needed refuge to birds, mammals, and other biota. Because they typically occur at points of ground-water discharge, and reduce the rate but prolong the duration of ground-water discharge, wetlands also provide survival areas to aquatic species sensitive to high or low temperatures during hot summer periods or cold winter periods. In addition, a wetland may offer the only wet habitat available during periods of prolonged drought or during fires to protect those biota able to seek refuge within it.

Wetlands function as an integral component of the local watershed. They tend to be highly productive areas, often serving as a source of organic detritus to adjacent water bodies. Wetlands also provide nursery areas for salmon and habitat for organisms that provide food to salmon and associated biota. The wetland contributes to the ecological balance within the watershed/ecosystem within which it occurs. Destroying, draining, or otherwise impairing the wetland's function alters the hydrologic, sediment, chemical, and biological balance in the watershed.

6.11 Salmonid Harvest

Although this document focuses on the effects of human activities on salmonid habitats, it is essential to recognize the effects fisheries have had on salmonid populations in the Pacific Northwest. The harvest of salmonids by humans constitutes a significant source of mortality for both anadromous and resident species. For thousands of years preceding settlement of the West by Euro-Americans, Native Americans depended on salmonids as an important source of food, and salmonids continue to be central to the culture and economy of many tribes. Since the mid-1800s, large number of salmonids have been taken in off-shore and in-river commercial and recreational fisheries. In the late 1800s and early 1900s, chinook salmon dominated commercial landings off Washington, Oregon, and California (Deimling and Liss 1994), as well as in-river fisheries in the lower reaches of the Columbia, Sacramento-San Joaquin, and other large river systems. These fish were targeted for their large size and high food quality. Subsequently, salmonid fisheries have become progressively more diverse with other anadromous salmonids, particularly coho and pink salmon and steelhead trout, accounting for growing fractions of the total catch.

The relative importance of different fisheries varies across the region. From central California to Cape Flattery, Washington, ocean commercial troll and recreational fisheries account for the highest catch of anadromous salmonids, although substantial in-river sport and tribal harvest occurs in some river systems. The Pacific Fishery Management Council (PFMC 1995) estimates that between 1971 and 1990, combined catch of coho salmon in the commercial troll and recreational fisheries off Washington (Puget Sound included), Oregon, and California averaged over 1.9 million fish annually. Catch of chinook in the region during the same period averaged 1.3 million fish, and average catch of pink salmon in odd-numbered years (primarily in Washington) was about 200,000 fish. The Oregon Department of Fish and Wildlife (ODFW 1982) estimates that from 1971–1975 the commercial troll fishery accounted for 67% of the total coho salmon harvest in the Oregon Production Index (OPI) area (Columbia River south to central California), with the ocean recreational fishery and Columbia River gill-net fisheries accounting for 23% and 8%, respectively. In-river recreational and tribal fisheries made up the remaining 2% of the catch. The allocation of chinook salmon among various fisheries in Oregon varies with region and life-history types. For north-migrating stocks, ODFW allocates approximately 50%–55% of total annual harvest to the ocean troll fishery and approximately 45% to the in-river recreational fishery. For south-migrating stocks, ocean troll fisheries account for 60%–67% of the total fish harvested, whereas in-river fishery targets are approximately 15%–37% of total harvest (ODFW 1991).

In the Puget Sound area (Strait of Juan de Fuca, Puget Sound, Hood Canal, Nooksack–Samish, Skagit, and Stillaguamish–Snohomishunits) commercial fisheries, including Indian and non-Indian, gill-net, purse seine, and troll fisheries, dominate the catch of salmonids: gill-nets and purse seines accounted for greater than 91% of the commercial catch in 1989 and 1990 (Palmisano et al. 1993a). Total commercial harvest in the Puget sound

area compares with the combined troll and recreational catch from coastal waters along Washington to California, with pink salmon and sockeye salmon being numerically dominant. From 1971–1990, combined annual catch of pink salmon (odd-years only) in treaty and nontreaty commercial fisheries averaged approximately 2.9 million fish, while catch of sockeye salmon averaged over 1.8 million. During this same period, catch of coho, chum, and chinook salmon averaged 972,000, 768,000, and 211,000 fish, respectively (PFMC 1993). From 1979–1990, sport harvest in the Puget Sound area averaged approximately 8.6% of the commercial catch (Palmisano et al. 1993a).

Estimating total harvest rates on specific stocks or species of anadromous salmonids is difficult. These calculations require accurate estimates of 1) total ocean and in-river harvest (including harvest of fish originating in Oregon, Washington, and California by fisheries in Alaska and British Columbia), 2) spawning escapement (sometimes direct counts, but often estimated from index streams), 3) indirect hooking and dropout mortality, and 4) rates of natural mortality for species with a variable period of ocean residence. Despite uncertainty associated with each of these estimates, calculations of total harvest rates for several anadromous salmonid populations provide some indication of the magnitude of fishing effects. Between 1960 and 1983, harvest rates of coho salmon in the OPI area ranged from 57% to 87%, with a mean exploitation rate of 71%. Harvest rates were lower from 1984–1993, ranging from 27%–62% with a mean of 43% (T. Nickelson, ODFW, personal communication). Further reductions in harvest rates occurred in 1994 and 1995 because of the closure of the coho fisheries. Ocean harvest rates of chinook salmon originating from the Sacramento-San Joaquin system in California ranged from 50%–79% between 1970 and 1992, with a mean harvest rate of 67% (PFMC 1993); in-river fisheries were not included in these estimates. In its management plan for coastal chinook salmon stocks, the Oregon Department of Fish and Wildlife (ODFW 1991) concludes that most chinook populations can sustain harvest rates below 67% without compromising long-term conservation goals. Actual target harvest rates for various stocks of chinook and coho salmon are adjusted depending on stock strength and specific escapement goals. Recent evidence indicates that biased selection of index streams has resulted in overestimation of spawning escapement and, hence, underestimation of harvest rates (Cooney and Jacobs 1994). In addition, harvest targets consider only numerical abundance and do not address other long-term effects discussed below.

Adverse effects of harvest on salmonids are particularly difficult to control in mixed-stock fisheries, where multiple species, stocks, and age classes are harvested together. Mixed-stock fisheries occur primarily in the ocean and lower river reaches, before stocks segregate into discrete spawning runs. Mixed-stock fisheries are difficult to manage because escapement goals and harvest rates of different stocks constituting the fishery cannot be controlled. Consequently, strong and weak stocks are harvested at comparable rates, as are fish of wild and hatchery origin. For example, in the Columbia River system, where 90%–95% of the coho salmon are of hatchery origin, harvest rates approaching 90% still allow adequate escapement for hatchery brood-stock purposes, whereas ODFW (1982) estimates that harvest rates should be less than 69% to meet escapement goals for wild coho. Thus, where hatchery and wild coho salmon commingle in the ocean, wild fish are likely to be harvested at an excessive rate. Mixed-stock fisheries are especially detrimental to naturally small populations or populations that have been depressed by human activities. In these populations, escapement may be insufficient to maintain genetic diversity, and the probability of undesirable founder effects increases.

In addition to reducing total escapement of adult salmonids, harvest alters the age- and size-structure of salmonid populations. For example, Ricker (1981) provided evidence that mean sizes of all five Pacific salmon species harvested in British Columbia have decreased over the past 30 to 60 years. Similarly, between 1935 and 1989, the average weight of coho salmon caught in commercial fisheries off the coast of Washington declined by almost 30%; over the same period, mean weight of chinook, pink, and sockeye salmon decreased by 24%, 19%, and 14%, respectively (Palmisano et al. 1993a). Changes in size and age-structure arise for several reasons. For long-lived species that spend several years at sea, such as chinook salmon and steelhead trout, decreases in average size and age occur because immature individuals are harvested by troll fisheries over a number of years. Thus, larger and older individuals are harvested at a higher rate than individuals that mature earlier and at smaller size (Moussalli and Hilborn 1986), particularly for those stocks frequenting coastal waters rather than only passing through coastal waters on their way to spawning areas. Changes in size structure may also result from size-selective fishing gear. Ricker (1981) attributed decreases in average size of coho and pink salmon adults to cumulative genetic effects caused by selective removal of larger individuals in troll and gill-net fisheries. Selective removal of larger fish may also increase the percentage of "jacks" in sockeye salmon populations. In freshwater fisheries, size limits and gear restrictions also alter size and age-structure of salmonid populations. For example,

Gresswell and Varley (1988) reported that mean length of cutthroat trout caught in Yellowstone Lake rose from a low of 365 mm in 1966–1967 to a high of 395 mm in 1983–1984 following a change in fishing restrictions from three fish of any size to two fish with a 330 mm maximum. Average age of spawners in Clear Creek, a major spawning tributary of the lake, increased from 4 years to almost 5.5 years during this period, with a concomitant increase in the frequency of fish aged 7–9.

Changes in average size and age of individuals influences success of salmonid populations in several ways. Large size provides salmonids with the energetic reserves needed to undertake extensive migrations as well as the ability to negotiate large barriers that are impassable to smaller fish; thus, the elimination of large individuals through harvest can effectively diminish the ability of populations to use particular spawning habitats. In addition, because the fecundity of salmonids typically increases with size, the selective removal of larger fish results in fewer eggs laid and ultimately a lower juvenile run than for a harvest pattern taking the same number of adults but no size selection (except possibly for populations that exceed the carrying capacity of their habitat) (Ricker 1972; Jaenicke and Celewycz 1994). Larger females also tend to dig deeper redds than smaller females, which reduces the likelihood that eggs will be destroyed by bedload movement during freshets. Larger females also select nest locations with larger gravel, which increases exchange of water and oxygen. Both of these behaviors combine to provide a greater egg-to-smolt survival (Hankin and Healey 1986; Hankin et al. 1993).

Harvest of salmonids also influences the timing of certain life history events, including adult migrations, spawning, and juvenile migrations. Frequently, fisheries are restricted to a relatively narrow window of time, particularly as stocks dwindle in numbers. Selective removal of early or late migrants can potentially result in shifts in the timing of peak migration and spawning within a population. Studies indicate that disproportionate representation of early migrants in hatchery broodstocks can cause a shift in migration timing within only a few generations (Millenbach 1973; Alexandersdottir 1987); harvesting only at the beginning or end of a run may have similar effects. Gharrett and Smoker (1993) reported that early and late-migrating adult pink salmon produce young that migrate to sea at different times. Consequently, removal of predominately early or late migrants can also alter the migration characteristics of the juvenile population.

Finally, the harvest of salmonids by humans can fundamentally alter the structure of stream ecosystems through reduction of nutrient inputs from salmon carcasses as populations decline and average size of fish decreases. Carcasses contribute significant amounts of nitrogen and phosphorus compounds to headwater streams (Cederholm and Peterson 1985; Bilby et al. 1996), the nutrients that most often limit production in oligotrophic systems. The role of carcasses in providing nutrients to stream systems is discussed in greater detail in Section 3.8.2.

6.12 Fish Introductions and Hatchery Management

Throughout history, humans have introduced fish into streams, rivers, and lakes in order to increase commercial and recreational fishing opportunities. These introductions have included both non-native species, primarily from the eastern United States and Europe, and artificially propagated native salmonids.

6.12.1 Introductions of Non-native Species

Introduction of non-native fishes into waters of the Pacific Northwest began before the turn of the century and continues today. Four primary sources of introductions include fishery management manipulations (stocking of fish); intentional introductions of gamefish by anglers; intentional or unintentional baitfish liberation by anglers; and bilge pumping of ballast water, particularly in estuaries and large rivers. Although there are few studies documenting conditions both before and after species introductions, effects of introductions on native fishes may include elimination, reduced growth and survival, and changes in community structure. For example, brown trout (*Salmo trutta*) replaced brook trout in a Minnesota stream over 15 years (Waters 1983), and cutthroat trout were replaced by more aggressive rainbow trout and brown trout in the Great Basin of western North America (Moyle and Vondracek 1985). Redside shiner were found to compete with young rainbow trout in Paul Lake, British Columbia, leading to decreased growth and survival of the young trout (Johannes and Larkin 1961). Ratliff and Howell (1992) reported that for 65 bull trout populations in Oregon considered at risk of extinction or already extinct, brook trout were the most important stressor in 26% of those populations and a contributing factor in 22%.

Moyle et al. (1986) identified six mechanisms that allow introduced fish to dominate or displace native fish including competition, predation, inhibition of reproduction, environmental modification, transfer of new parasites or diseases, and hybridization. They suggest that introduced species may thrive best where extensive environmental modification has already occurred. In the Columbia river—a system where temperature and stream velocities have been substantially altered—predator species introduced for recreational fishing, including walleye *(Stizostedion vitreum),*

channel catfish (*Ictalurus punctatus*), and smallmouth bass *(Micropterus dolomieui),* are feeding on outmigrating smolts (Palmisano et al. 1993a). Introduced grass carp (*Ctanopharyngodon idella*) and common carp (*Cyprinus carpio*) destroy beds of aquatic macrophytes, which reduces cover for juvenile fishes, destroys complex substrates that support diverse invertebrate assemblages, and increase the turbidity of water to the detriment of fishes that locate prey by sight (Moyle et al. 1986). Whirling disease—which was introduced to North American waters from Europe via shipments of frozen fish containing spores of *Myxobolus cerebralis* (Marnell 1986)—has been implicated in the decline of several important trout fisheries in the intermountain West, particularly in the upper Colorado River basin (Nehring and Walker 1996; Vincent 1996). Although this disease has been found in hatcheries within the Pacific Northwest, there is little evidence that it has affected wild trout populations in the region (Nehring and Walker 1996). Cutthroat and rainbow trout freely hybridize in the wild, with the rainbow trout phenotype becoming dominant (Behnke and Zarn 1976); similarly, the various subspecies of these trouts also interbreed.

6.12.2 Artificial Propagation of Native Salmonids

Artificial propagation of native salmonids has been used for decades as a means of replacing lost natural production resulting from various development activities and to increase returns for harvest. Hatchery programs continue to dominate expenditures of State fishery agencies in the Pacific Northwest. White et al. (1995) report that the State of Washington spent $31.3 million—35% of their total fishery budget—on salmon culture in the 1991–1992 fiscal year. Similarly, propagation of fish accounted for 42.5% of Oregon's $90.6 million budget for fisheries for the 1993–1995 biennium, whereas only 3% was devoted to management for natural production.

Although artificial propagation may in some instances increase salmon and trout available for harvest, hatchery introductions can result in a number of unintended and undesirable consequences for wild salmon and trout populations (Lichatowich and McIntyre 1987; White et al. 1995). In freshwater, interactions between hatchery and natural fishes may result in greater competition for food, habitat, or mates (Nickelson et al. 1986). Studies have suggested that carrying capacity can be exceeded during the outmigration of smolts to the ocean (Steward and Bjornn 1990). Once in the ocean, large numbers of hatchery smolts may result in density-dependent decreases in survival and growth, although evidence of density-dependent effects in ocean environments is mixed. Indications of density-dependent age and size composition have been found for various Pacific salmonids (Ricker 1981; Peterman 1987; Ishida et al. 1993). Peterman (1978) found that only a few salmonid stocks exhibit density-dependent marine survival patterns, and that these effects were predominantly within or between cohorts—little or no marine density-dependence was found between different stocks, whether derived from nearby or distant spawning areas. Holtby et al. (1990) found no evidence for density-dependent marine survival of coho salmon migrating from Carnation Creek, British Columbia. It is possible that density-dependent ocean survival may only be manifest in years of low marine productivity.

Other adverse effects of hatchery introductions include transmission of disease between hatchery and wild populations (Marnell 1986; Steward and Bjornn 1990), alterations of fish behavior, and increased predation on wild fish. In 1987, the incidence of BKD infection in hatchery spring chinook from two Snake River hatcheries was 92% to 99% (Bevan et al. 1994a). Because many fish may carry BKD for extended periods without exhibiting symptoms, cross-transmission may be substantial. The likelihood of transmission may be particularly high when fish are aggregated for transport in raceways, trucks, and barges. Alteration of behavior of wild fish, including stimulation of early migration of juveniles (Hillman and Mullen 1989), has been observed in response to hatchery introductions. Hatchery supplementation can also increase predation rates on wild stocks either directly, through predation of hatchery fish on wild fish, or indirectly by attracting predators.

In addition to ecological effects, introduction of hatchery fish may lead to genetic changes in wild populations (Hindar et al. 1991; Waples 1991a). Introduction of hatchery stocks can eliminate unique genomes in local stocks. Straying and subsequent crossbreeding may result in loss of genetic variability between populations and depressed fitness where introgression occurs. Low rates of natural straying may be beneficial in maintaining genetic variability in natural populations, but these rates may become elevated through artificial propagation (Bams 1976; Withler 1982), with potentially serious consequences for locally adapted populations.

The operation of hatchery facilities may adversely affect wild salmonid populations and their habitats in several ways (reviewed in White et al. 1995). Effluent waters from hatcheries may contain high concentrations of nutrients or disinfectant chemicals that negatively affect water quality. Disease organisms can also be introduced to streams via hatchery effluent. The construction of hatchery weirs or diversion structures impedes the migration of wild stocks and diversions of water for hatchery use reduces the amount available for wild stocks. Removal of wild fish for brood stock may threaten

the genetic integrity of wild stocks, particularly for small or depleted stocks. And lastly, the removal of fish for brood stock decrease the amount of nutrients available in upstream reaches, since salmon carcasses are not deposited.

Hatchery supplementation has social repercussions that influence wild salmonids directly, as well as the ability of managers to restore salmonid populations. Hatchery supplementation increases harvest pressure on wild populations in mixed-stock and terminal fisheries ((Palmisano et al. 1993a; Lichatowich and McIntyre 1987), particularly during years when survival of hatchery fish is low due to poor environmental conditions, For example, the overcapitalization of the coho salmon fishery and subsequent overharvest of wild stocks in Oregon in the late-1970s and 1980s was stimulated in part by successful hatchery supplementation during the 1960s and early 1970s (Lichatowich and McIntyre 1987). In addition, once commercial and sport fishers have invested large sums of money in fishing boats and gear, they may become resistant to increased fishing restrictions, making it difficult for managers to enact stricter protections for wild stocks.

Finally, the long history of hatchery programs in the United States has instilled a perception in the public that habitat losses or degradation can be mitigated through artificial propagation (White et al. 1995), or that maintenance of salmon populations depends on hatcheries (Hilborn 1992). The disproportionate spending of State and Federal dollars on hatchery programs compared with protection of natural habitat and wild populations is indicative of the reliance that the public places on artificial propagation. As White et al. (1995) point out, political pressure for stocking has driven management decisions even in cases where scientific evidence has indicated stocking is not needed or detrimental. This pressure has also diverted much-needed funds from other important and more ecologically sound restoration activities.

6.13 Recreation

Although the primary influence of recreation on salmonids is fishing, there are also indirect effects related to boating, log removal, parks, and campgrounds. Stream and lake banks, riparian vegetation, and spawning redds are disturbed wherever human use is concentrated (Johnson and Carothers 1982); however, these effects are generally localized. Human concentrations at campgrounds or vacation areas may also lead to impaired water quality by elevating coliform bacteria and nutrients in streams (Aukerman and Springer 1976; Potter et al. 1984). Recreational boaters, kayakers, and rafters have less obvious, but more far-reaching effects, by removing snags from rivers and lakes. This is done for reasons of aesthetics and safety, but popular whitewater rivers and many recreational lakes are nearly devoid of snags. Removal of this wood potentially affects salmonids by reducing habitat complexity in rivers and in estuaries into which they enter. The reduced number of logs lowers estuarine and marine habitat quality for fishes just as it does for habitat in rivers (Maser and Sedell 1994).

6.14 Beaver Trapping

Other than humans, the mammal that most shaped North American waterways was probably the beaver. In pre-Columbian times, their numbers were estimated to be 4–26 km^{-2} across the United States (Naiman et al. 1986), and they provided the initial economic base for European exploration and settlement west of the Appalachians. However, because of widespread trapping in the 1800s and early 1900s, their numbers have dwindled to a fraction of their historical abundance (0.4–0.8 km^{-2} today (Naiman et al. 1986). Beavers have both negative and positive effects on water bodies and riparian ecosystems. Their feeding results in the loss of woody riparian vegetation and increased retention of fine sediments, but increases the input of large woody debris to streams. Beaver ponds increase the surface-to-volume ratio of the impounded area, thereby increasing summer temperatures. Marcus et al. (1990) suggest that in the east, temperature increases may be detrimental to trout populations, but that in the Rocky Mountains, increased temperature where waters are colder, may benefit salmonids. Beaver ponds also supplement summer low flows (Marcus et al. 1990) and provide critical over-wintering habitat for salmonids. Bank dens and channels increase erosion potential, but also offer juvenile salmonids protection from high winter flows. Beaver ponds frequently fill with sediments to become wetlands, but they retard erosion upstream and reduce sedimentation downstream. A high frequency of ponds may reduce the amount of spawning gravel through siltation (Marcus et al. 1990). When channels are once again established, these reaches produce large numbers of fish. Beaver ponds in the Rocky Mountain West were found to support larger and more numerous trout, as well as greater densities of aquatic invertebrates than undammed sections of the stream (Naiman et al. 1984). Beaver ponds may also provide a sink for nutrients from tributary streams, enhancing pond productivity, and increasing retention time (Maret et al. 1987; Naiman et al. 1986). While it is difficult to generalize about the overall effect of beaver on salmonids, Naiman et al. (1986) suggested beaver act as a keystone species "to affect ecosystem structure and dynamics far beyond their immediate requirements for food and space." Their removal has fundamentally altered aquatic ecosystem function.

7 Oceanic and Atmospheric Circulation

Until the mid-1970s, little was known about the effect of oceanic conditions on anadromous salmonids. Most research on salmonid biology focused on the freshwater environment, and fishery biologists generally attributed variation in population size to conditions in fresh water. Recent work strongly suggests that marine productivity depends on atmospheric and oceanic circulation patterns and that the abundance of salmonids and other fishes may be greatly affected by short- and long-term variation in those patterns (Mysak 1986; Roesler and Chelton 1987; Francis and Sibley 1991; Ware and Thomson 1991). Growing evidence suggests that conditions in the northeastern Pacific Ocean shifted abruptly in the mid-1970s and that salmonid populations along the entire western coast of North America have responded to these large-scale changes (Francis and Sibley 1991; Pearcy 1992). It is clear that efforts to restore freshwater habitats of salmonids need to be considered in the context of larger-scale fluctuations in numbers brought on by climatic and oceanic conditions.

In this chapter we briefly review general circulation patterns and the dominant physical processes controlling conditions in the northeastern Pacific Ocean. We then present hypothesized mechanisms by which salmonid abundance and life histories may be influenced by changing oceanic conditions. Finally, we discuss implications of long-term variability in marine conditions for strategies to restore salmonids in the Pacific Northwest.

7.1 General Ocean Circulation

Circulation in the northeastern Pacific is dominated by the behavior of the Subarctic Current and the West Wind Drift, large west-to-east surface currents situated at approximately 42–49° north latitude. These currents bifurcate as they approach North America with the Alaska Current flowing north and the California Current flowing south (Figure 7-1) (Ware and McFarlane 1989). These surface currents interact with prevailing wind patterns and the rotation of the earth to produce distinct upwelling and downwelling patterns along

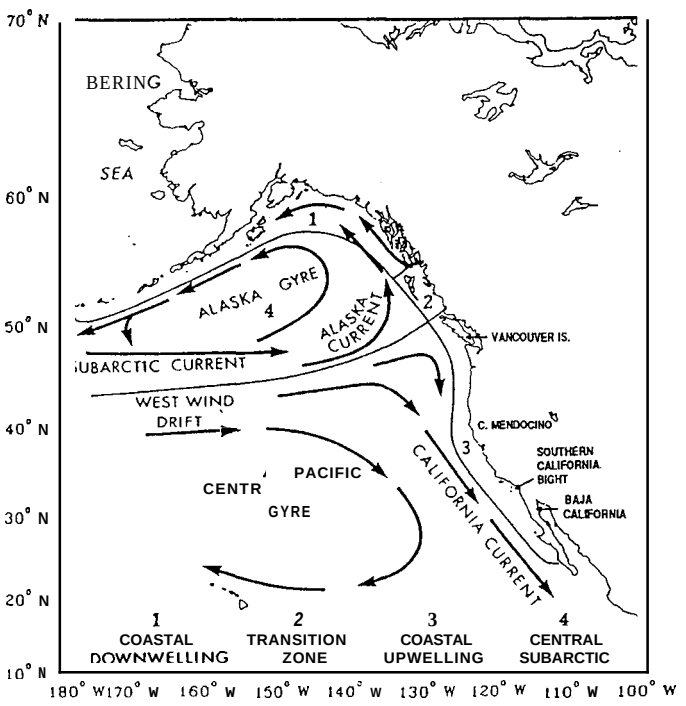


Figure 7-1. Approximate areas of oceanic domains and prevailing current directions in the northeastern Pacific Ocean. From Ware and McFarlane *(1989).* Reproduced with permission from the publisher.

different parts of the North American coast. Wind blowing across the ocean surface causes displacement of surface waters at an angle 90° to the right of the direction of the wind. South of Vancouver Island, BC, northwesterly winds generally blow along shore from May to September, causing surface waters to be transported offshore and resulting in the upwelling of cold water along the continental margin (Ware and McFarlane 1989). North of Vancouver Island, the movement of surface water is directed generally toward the shore by prevailing winds, and downwelling persists for most of the year (Ware and McFarlane 1989).

The behavior of this large-scale oceanic circulation varies from year to year and at longer time scales, depending on atmospheric conditions and particularly on the strength of the Aleutian low-pressure system off the coast of Alaska. In years when the Aleutian Low is well-developed, the position of the Subarctic Current shifts to the south (Mysak 1986), and a greater proportion of water in the Subarctic Current and the West Wind Drift is diverted northward to the Alaska Current (Pearcy 1992). These conditions are characteristic of El Niiio years, when warm waters from the subtropics shift to the north. Conversely, when the Aleutian Low is poorly developed, the Subarctic Current shifts to the north, and the California Current receives a higher fraction of the total water. Ware and Thomson (1991) have proposed that long-period oscillations (40–60 years) in wind-induced upwelling significantly influence oceanic conditions along the coast. Thus, short-term variations in the strength of coastal upwelling and the occurrence of El Niiio events appear to be overlaid on oscillations of longer periodicity associated with atmospheric circulation.

7.2 Ocean Conditions and Salmonid Production

Variation in oceanic circulation patterns along the North American coast greatly affects characteristics of seawater, including surface-water temperatures, salinity, sea-level height, and nutrient concentrations, which in turn affect the abundance and distribution of aquatic organisms. High temperatures, reduced upwelling, and inshore depression of the thermocline during the strong El Niiio event of 1982–1983 together resulted in significant declines in phytoplankton production along the coast of Oregon (reviewed in Pearcy 1992). Similarly, zooplankton biomass was greatly reduced and species composition **shifted** to **taxa** more commonly found in more southern waters. Roesler and Chelton (1987) attributed interannual differences in the biomass of zooplankton in spring off the coast of north-central California to differences in advective transport of zooplankton from arctic waters by the California Current. Thus, changing ocean conditions can significantly affect the amount of food available to juvenile salmonids as they enter the ocean.

In addition to affecting food supply, changing oceanic conditions may also affect the distribution and abundance of predators and competitors. Holtby et al. (1990) speculated that warmer ocean temperatures off the coast of Vancouver Island may lead to northward shift in populations of large piscivorous predators such as Pacific hake (*Merluccius productus*). Increases in predator abundance and concomitant decreases in alternative prey species (e.g., Pacific herring [*Clupea pallasi*]) may result in greater mortality to salmonids off the coasts of California, Oregon, and Washington during El Niño years. Another hypothesis attributes fluctuations in ocean survival of salmonids to changes in the off-shore transport of juveniles as they enter the ocean. During years of high upwelling, smolts may be transported off shore where they are less vulnerable to sea birds and other predators that are abundant along the coast line, whereas in years of poor upwelling salmonids may remain in near-shore areas (reviewed in Pearcy 1992). Migration routes of juvenile sockeye salmon from the Fraser River differ substantially in El Niño versus La Niiia years (Mysak 1986).

Regardless of the specific mechanisms controlling salmonid abundance in the ocean, the evidence is clear that oceanic conditions play a significant role in regulating survival. Numerous studies have linked marine survival of coho salmon in the ocean with the strength of upwelling (Gunsolus 1978; Nickelson 1986; Fisher and Pearcy 1988). Francis and Sibley (1991) demonstrated long-term fluctuations in the catch of coho salmon off the coast of Washington, Oregon, and California, which they attributed to changes in the marine environment caused by climatic change. Interestingly, pink salmon catches in Alaska have oscillated out of phase with coho salmon in the Oregon Production Area, indicating that conditions leading to high production of salmonids in the Coastal Upwelling Domain have adverse affects on salmon in the Coastal Downwelling Domain and vice versa (Francis and Sibley 1991).

In addition to affecting the survival and productivity of salmonids in the northeastern Pacific, variability in marine conditions has likely influenced the evolution of life history characteristics of salmonids (Holtby et al. 1989). Spence (1995) examined migration timing of coho salmon smolts from 50 populations along the coast of North America and found distinct regional differences in migration characteristics. Coho populations in the northern part of the range typically migrate during a relatively short and predictable period during the late

spring. In contrast, southern populations generally exhibit a more protracted migration that peaks earlier in the spring but is more variable from year to year. Spence (1995) suggests that, in part, these differences likely reflect the adaptation of populations to differences in the degree of predictability in oceanic conditions in the northern and southern parts of the coho salmon's range. **As** knowledge of the marine ecology of salmonids increases, additional patterns in life-history characteristics of salmonids will undoubtedly emerge.

7.3 Implications for Restoration

Cycles in marine productivity have the potential to mask the effects of degradation in freshwater habitats. Lawson (1993) presented a conceptual model for considering the combined effects of oceanic cycles and habitat degradation in fresh waters (Figure 7-2). **As** freshwater habitats are degraded, salmon populations do not decline in linear fashion. Instead, a general downward trend is masked by long-term oscillations in ocean productivity. During periods of unfavorable ocean conditions, the consequences of degradation in freshwater habitats become most evident, and the risk of local population extinction becomes greatest. **As** Lawson (1993) pointed out, there may be a tendency for fishery managers and politicians to relax as populations begin to recover—which they eventually will do provided they do not go extinct during a poor ocean phase—even though the quality of freshwater habitats continues to decline. Similarly, ill-conceived restoration strategies may appear to be successful as salmonid numbers increase, even though those increases are merely the fortuitous result of improving oceanic conditions.

Long-term oscillations in ocean productivity also have a significant bearing on harvest and hatchery management. Harvest projections and limits typically are based on maximum sustained-yield models that assume a constant environment. Over the term of their prediction, these models assume linear relationships between production and yield. Such models are particularly problematic in a changing environment or in one that is tending in a direction different from that in which the model was developed. Similarly, the survival and production of hatchery fish may vary significantly with conditions at sea (Pearcy 1992). In the 1950s and 1960s coho salmon hatcheries in Oregon were enthusiastically endorsed by fishery managers and commercial fishers (who **tripled** in number over a 10-year period) because of early success that was largely the result of favorable oceanic conditions. When environmental conditions shifted in the mid-1970s, survival of hatchery coho decreased, and the overcapitalized

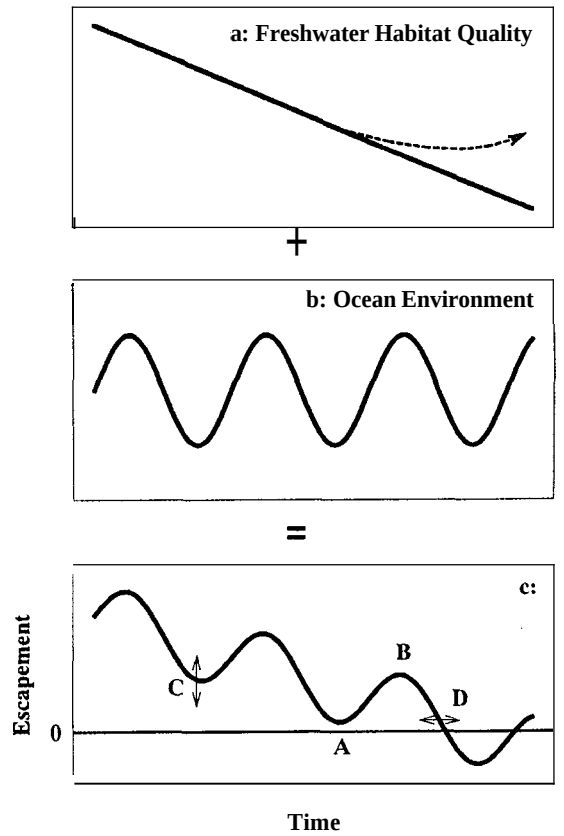


Figure 7-2. Conceptual model of effects of declining habitat quality and cyclic changes in ocean productivity on the abundance of Oregon's coastal natural coho salmon. For the labels: "a" indicates trajectory over time of habitat quality (dotted line represents possible effects of habitat restoration); "b" shows generalized time series of ocean productivity: "c" is the sum of top two panels where A = current situation, B = situation in the future, C = change in escapement from increasing or decreasing harvest, and D = change in time of extinction from increasing or decreasing harvest. From Lawson (1993). Reproduced with permission from the publisher.

fishery took an increasing toll on wild stocks (Pearcy 1992). Significant economic hardship for coastal communities and precipitous decline in wild coho populations resulted.

Lawson (1993) concludes that, in the face of natural variation in ocean productivity, salmonid restoration should proceed in three phases: 1) short-term projects, 2) long-term projects, and 3) monitoring. Short-term projects should be directed at immediate and readily identifiable habitat problems where manipulation can temporarily enhance production (e.g., creation of off-channel pools, cleaning of gravels). Long-term projects should be directed at restoring natural ecological processes, and include such things as replanting of riparian zones or

re-establishment of wetlands. Monitoring is essential to ensure that both short- and long-term projects are effective, but the metrics used to gauge success may be substantially different. Using numerical abundance of salmonids (particularly juveniles) to measure success has pitfalls within a highly fluctuating ocean environment. The focus should be whether restoration strategies are effective over decades or even centuries, not years. For long-term monitoring, indicators should measure restoration of ecological functions or processes rather than merely record fish abundance.

8 Practices to Restore and Protect Salmonids

In preceding chapters, we have discussed a wide array of effects that human activities have on aquatic salmonids, their habitats, and aquatic ecosystems in general. In this chapter, we review various practices and programs by which these effects can be avoided or substantially reduced. We first discuss practices that directly influence fish (harvest and hatcheries), then move to water- and land-use practices that directly or indirectly influence the integrity of water bodies (waterway modification, forestry, grazing, agriculture, mining, and urbanization). We conclude with a discussion of broader societal values and policies that drive resource consumption and, hence, determine the health of ecological systems. It is important to note that most programs intended to protect or restore streams have focused primarily on one problem area (e.g., stream bank, stream reach) and one stressor (e.g., grazing, agriculture, forestry, fish harvest) but have not tried to integrate other stressors into a comprehensive solution for the watershed. Although various practices and programs are discussed separately, we emphasize the need to integrate them into an ecosystem management approach at multiple spatial scales from sites and watersheds to basins and regions (see Part II of this document). In addition, regardless of the land-use activity, emphasis should be placed on preventing (rather than mitigating) damage, particularly in those areas where high-quality habitats and stable salmonid populations remain (Frissell et al. 1993; Bradbury et al. 1995).

8.1 Harvest Management

As discussed in Section 6.11, harvest of salmonids can both reduce total escapement of adult salmonids and alter the age- and size-structure of salmonid populations (Ricker 1981). Size-selective gear coupled with high rates of harvest for older age classes typically result in shifts towards younger, smaller adults. In mixed-stock fisheries where weak stocks are harvested at the same rate as more abundant stocks (including hatchery stocks), these effects can be exacerbated.

The NMFS Snake River Salmon Recovery Team recommended terminal area fishing and selective fishing as the best harvest schemes where mixed-stock fisheries include weak, depressed, or endangered stocks (Bevan et al 1994a, 1994b, 1994c). Terminal and bay fisheries (i.e., fisheries that target adults as they return to their natal streams) provide greater protection for weak stocks by targeting hatchery runs instead of wild stocks, by allowing late-maturing fish like chinook salmon to reach maturity, and by reducing the incidental mortality of subadults. Such systems are easily opened and closed to increase escapement at particular times during the run and have been employed successfully in many regions. Secondary benefits of shifting from open ocean to near-shore fisheries include reduced costs and lower risks associated with fishing.

In contrast to terminal fisheries, the management of open-ocean harvest requires a greater understanding of the distributions of different stocks in order to determine their vulnerability to the fishing fleets. The presence of stocks of differing degrees of robustness can be dealt with by managing for indicator stocks or for weak stocks. The use of indicator stocks presents the risk that the chosen indicators are not truly representative of all stocks in the fishery. Serious harm to more fragile stocks can occur when healthier indicator stocks are performing well. Conversely, potential harvest from healthy stocks may be lost by managing strictly for the weakest stocks (Kope 1992; Restrepo et al. 1992). Weak stock management has the latter risk (higher levels of yield foregone) and the added difficulty that smaller populations tend to have less data available concerning their production and recruitment mechanisms, as well as fishery impacts (Peterman 1978; Peterman and Steer 1981; Lestelle and Gilbertson 1993). In practice, a mixture of the two techniques is usually attempted, managing for indicator stocks or complexes of stocks while also including attention to weaker stocks vulnerable to the fishery.

Harvest methods can also be changed to target hatchery stocks and reduce incidental mortality of wild populations. Wild fish caught in traps and fish wheels or by hook and line (especially with lures or barbless hooks) are less likely to die than those captured with gillnets or by trolling. Special sport-angling restrictions, including catch-and-release angling, minimum size or slot limits, and bag limits

may further reduce mortality or minimize size-selective harvest. Accurate monitoring of escapement levels of specific stocks is essential for establishing exploitation levels that ensure the long-term persistence of individual stocks (e.g., Cooney and Jacobs 1994).

8.2 Hatchery Practices

Growing evidence of adverse ecological, genetic, and social consequences of hatchery operations (reviewed in Section 6.12) has prompted many fishery scientists to recommend substantial modification, curtailment, or elimination of hatchery programs for salmonids (Lichatowich and McIntyre 1987; Waples 1991b; Hilborn 1992; Meffe 1992; Bevan et al. 1994a; White et al. 1995). Reductions in hatchery production would disrupt harvest levels over the short term. Nevertheless, competition, predation, disease transmission, and genetic introgression resulting from fish stocking threaten the persistence of wild salmonids, particularly those populations that have been depleted because of habitat degradation and overharvest.

White et al. (1995) recommended that fish stocking be limited to three kinds of temporary programs and two types of prolonged programs. They suggested that temporary stocking should be used only to 1) recolonize native species into waters where fish have been extirpated by human activities, after those activities (e.g., habitat destruction, introduction of non-native species, overharvest) have been reduced or eliminated; 2) create populations in new artificial waters (e.g., constructed lakes or ponds), and 3) sustain a presently overharvested fishery through a planned program of downsizing and transition to other employment or from reliance on hatchery fish to reliance on wild fish. Prolonged programs they recommended include hatcheries to 1) augment weak stocks (put-and-grow stocking) in waters having little or no reproductive habitat but substantial productive potential, but only where it will not harm indigenous biota and 2) provide highly artificial opportunities for recreation where fishing will be intensive (e.g., put-and-take stocking of catchable-sized fish, for quick and easy catch in urban ponds). White et al. (1995) argue that supplementation stocking (defined as increasing natural production) is "out of keeping with ecological reality and rational management." Although certain societal values may be achieved with hatcheries (Schramm and Mudrak 1994), these values can often be attained through angling restrictions and restoration of wild populations without hatchery supplementation.

The issue of hatchery supplementation is particularly contentious as it relates to augmenting threatened and endangered stocks through the use of wild brood stock from the endangered population. In deciding whether to use artificial propagation to conserve endangered species, a key consideration is the likelihood that such efforts will actually benefit the listed species (Waples 1991b). Although artificial propagation of Pacific salmon has been carried out on a large scale for many decades, almost all these efforts have been directed at enhancing fisheries harvest. Attempts to increase natural production through the use of artificial propagation is a relatively recent enterprise that has, to date, produced mixed results (Miller et al. 1990). Removal of threatened and endangered species for use as brood stock may increase the genetic vulnerability of the population and hence its long-term prospects for survival. For example, a recent modeling exercise by Currens and Busack (1995) indicated that the risk of extinction for a chinook salmon population was greater with supplementation than without it. Consequently, the use of artificial propagation to conserve listed species should be viewed as experimental and highly risky given the historical effects of hatcheries on reducing the biological diversity of salmonids (Bottom 1997).

One of the most disruptive influences of hatcheries has been the introduction of non-native species to Pacific Northwest waters. These species include competitors, such as brook trout and American shad, as well as several large piscivores including walleye, smallmouth bass, largemouth bass (*Micropterus salmoides*), and striped bass (*Morone saxatilis*). Obviously, the most direct way to minimize these impacts is to cease stocking both waters that contain wild salmonids with non-native species and waters from which non-natives can disperse into salmonid-bearing streams. Other management strategies that encourage elimination of non-native fishes include direct removal by piscicides and electrofishing, and indirect removal through use of unrestricted catch limits (Bevan et al. 1994a). Restoring streams and rivers to their natural temperature and flow regimes may reduce the spread of non-native species into salmonid streams. For example, evidence from California suggests that streams with natural flow regimes are less prone to invasion by non-native species than highly regulated streams where seasonal flow fluctuations have been artificially dampened (Baltz and Moyle 1993). Finally, it is important to note that many State and Federal agencies continue to promote the stocking of non-native species into certain waters, although new introductions have been curtailed in recent years as understanding of the ecological consequences of species introductions has increased. Stocking programs both contribute directly to declines in native salmonids (and other native biota) and create social constraints that may impede restoration of wild

fish (e.g., sport angling constituencies that favor fisheries for non-native species).

8.3 Waterway Modification

Throughout human history, waterways have been used for water supply, waste discharge, power generation, and transportation—values that have collectively made waterways centers for development. With growth in human populations and commerce, many streams, rivers, and estuaries were increasingly simplified by dams, channelization, revetments, snagging, and removal of wetlands and side channels. Seasonal peak and low flows were stabilized by dams, although in systems with hydroelectric facilities, daily fluctuations were increased to meet power demands. Water quality deteriorated as a result of these changes combined with discharge of municipal and industrial wastes, as well as water withdrawals for industrial, agricultural, and domestic uses. Particularly damaging to aquatic life were increased temperature, turbidity, sedimentation, toxics, nutrients, and oxygen-demanding wastes. The progressive commercial, agricultural, and urban development stimulated additional alterations, often at increasingly greater distances. In places such as western Europe, where these changes have been taking place for centuries, natural channels and high water quality remain in only the most remote areas.

Substantial strides have been made toward improving the water quality of the Nation's surface waters. Many of the most extreme instances of pollution have been substantially reduced. Sensitive forms of aquatic life and recreational opportunities are returning to all but the worst of our water bodies, and the frequency of waterborne disease has been dramatically reduced. Nationally, these changes were stimulated and financed by the Federal Water Pollution Control Act (FWPCA) and subsequent amendments that form the Clean Water Act (CWA; see Chapter 9); however, several progressive States, as well as industries, preceded the FWPCA in cleaning up point-source discharges. Several lessons in waterway protection and restoration can be learned from the Federal water quality legislation, nonfederal responses, and related legislation: 1) a clear set of goals and objectives is needed; 2) funding must be set aside for applied research and training; 3) cost-sharing grants are often necessary to construct treatment works; 4) numerical criteria are required for various pollutants; 5) both site-specific and basin-wide permitting, monitoring, and reporting systems are useful for controlling pollution and evaluating compliance; 6) improved land-use can effectively control diffuse pollution; 7) ambient biological criteria and biological surveys are necessary for evaluating the biological effectiveness of controls; 8) reducing wastes frequently saves money for those who discharge pollutants; 9) the Federal government can provide overall guidance and direction, but States can effectively conduct the monitoring and enforce the regulations; 10) the effectiveness of pollution-reduction programs ultimately depends on the character and creativity of those discharging contaminants into rivers and streams.

Recently, researchers have identified the vital role of channel complexity, riparian zones, and floodplains in the productivity and diversity of the aquatic habitat and its organisms (Sedell and Luchessa 1982; Maser et al. 1988; Gregory et al. 1991; NRC 1992). Numerous programs are presently underway to increase river-floodplain interactions for the benefit of instream habitat and riparian zones. Approaches range from active natural channel restoration to nonstructural passive methods for floodplain protection. The Kissimmee River, Florida, restoration project has been called a model for watershed restoration (Niering and Allen 1995). It calls for physical reconfiguration of the river to re-establish backwaters and contact with the floodplains and for re-establishment of historical inflows from the river's upper watershed. A similar effort has been planned for portions of the Missouri River (Hesse and Sheets 1993; Galat and Rasmussen 1995). That approach includes an evaluation of predisturbance conditions with the intent of identifying areas where side channels, wetlands, and bottomland hardwood forests can be restored through the removal of revetments and set back of levees.

Other researchers have identified strategies to minimize adverse affects of channelization through stream renovation. McConnell et al. (1980) suggested that snagging only one side of a stream halves costs and deleterious impacts to the environment. Nunnally et al. (1978) proposed minimizing erosion and reducing hydraulic efficiency by maintaining natural meanders when channelizing streams. The more natural stream channels and riparian vegetation also improve biological and aesthetic benefits. Alternative strategies such as these involve significant environmental tradeoffs but are less destructive than traditional channelization.

Several proposals have been put forth to restore natural ecological processes in rivers by removing existing dams. The Maine Legislature requested that Edwards Dam be removed from the Kennebec River to improve migration of Atlantic salmon. Despite fish-passage modifications, the dam still blocks migrations of salmon as well as migrating sturgeon, shad, and smelt while providing electricity to fewer than 2,000 homes (NRC 1992). The Elwha River Ecosystem and Fisheries Restoration Act (Public Law 102-495) was passed in 1992 to remove two dams on

the Elwha system for the purpose of restoring the riverine ecosystem and historic runs of five Pacific salmon species, once among the most prolific runs on the Olympic peninsula of Washington (Wunderlich et al. 1994). The dams also are associated with the disappearance of 22 bird and mammal species from that area of Olympic National Park with consequent, annual recreational revenue losses of $500,000. Electricity from the dams is sold to a paper company, and replacement sources of power have been identified. An analysis determined that removal of both dams is the only option that will allow full restoration of the watershed (NRC 1992). In Oregon, two dams on the Rogue River have been identified for potential removal. The Bureau of Reclamation (BR 1995) estimated that removal of Savage Rapids Dam would increase escapement of adult salmon and steelhead by 26,700 fish, increase harvest by 87,900 fish, and cost taxpayers nearly $8,000,000 less than dam retention (with modifications for improved fish passage); these benefits to fisheries would be attained while still providing irrigative withdrawals, the primary function of the dam. Additional dam removals are called for throughout the Pacific Northwest because of inadequate or no fish passage, excessive sedimentation, structural deterioration, and hazardous or unsafe conditions.

A variety of programs have been aimed at maintaining the viability of fish stocks by mitigating the impacts of dams. These programs provide upstream passage for mature fish and downstream passage of smolts. The Northwest Power Planning Council's Strategy for Salmon identifies immediate actions to aid juvenile passage (NPPC 1992a, 1992b): screen all turbines, improve bypass systems, lower reservoir pools during smolt migration, barge past dams, place a bounty on predators, and boost flows during out-migration. The Snake River Recovery Team made similar final recommendations to NMFS (Bevan et al. 1994a, 1994b, 1994c).

Reintroduction of beaver has also been recommended as a means of restoring stream habitats. Beaver dams can increase habitat complexity, slow stream incision, and increase flows during the dry season by allowing greater subsurface storage of water in floodplain areas. Although beaver ponds may eliminate spawning areas, warm the water, and reduce dissolved oxygen, they create additional areas for rearing, over-wintering, escaping freshets, and trapping sediments. Their introduction to higher gradient salmonid streams is associated with higher salmon densities and significantly greater overwinter survival (Phillips 1987; Swanston 1991).

The Fish and Wildlife Service has developed the Instream Flow Incremental Methodology (IFIM) to provide a means of systematically evaluating alternative flows for the protection or enhancement of aquatic resources in regulated river systems (Armour and Taylor 1991). The IFIM couples models of physical habitat conditions at different stream discharges with information on specific habitat requirements of fish to estimate minimum instream flows needed below river impoundments. Maintaining instream flows is especially important for eastside streams where irrigation needs or hydroelectric operations may severely deplete base flows. Though widely used, a number of concerns of IFIM concepts have been expressed. Nestler (1990) summarized these concerns, arguing that depth, velocity, and substrate are inadequate estimates of habitat quality and that habitat quality is only a coarse estimate of population density. Stalnaker (1990) argued that IFIM has been used to establish minimum flows that are frequently violated and that ignore other necessary flows such as floods. Armour and Taylor (1991) pointed out that the methodology was designed to evaluate alternate flow regimes not as the definitive answer for flow disputes. The results simply provide a framework for negotiating flows to be maintained.

During the past two decades, increasing effort and resources have been committed to instream artificial structures intended to improve fish habitat. The National Research Council (1992) provided a summary of 22 habitat-improvement evaluations deemed successful based on increased fish density during the period evaluated. No time period for evaluation is given, nor are metrics provided that were used as the basis for the evaluation. The results from these studies should be interpreted with caution because they originated in systems east of the Rocky Mountains, systems having different climate, geology, sediment transport, hydrology, and gradient than Northwest streams. In contrast, Frissell and Nawa (1992) surveyed artificial structures in streams of western Oregon and Washington and concluded that "commonly prescribed structural modifications often are inappropriate and counterproductive." They reported frequent damage to artificial structures, particularly those located in low-gradient reaches and in streams with recent watershed disturbance. When evaluated for 5–10 year damage rates, overall median failure rate was found to be 14% and median damage rate (impairment plus failure) was 60%. They concluded that streams with high or elevated sediment loads, high peak flows, or highly erodible bank materials are not good candidates for structural modifications. Beschta et al. (1991) surveyed 16 stream-restoration projects in eastern Oregon and found that instream structures frequently had negative effects on aquatic habitats (e.g., altered natural biotic and fluvial processes), were inappropriate for the ecological setting (e.g., boulders or large wood placed in meadow systems that historically never had

such structures), or did not address the full suite of riparian functions that contribute to habitat quality. They concluded that in most instances instream structures are unwarranted and should be eliminated as a restoration method; re-establishment of riparian vegetation through corridor fencing or rest from grazing was determined to be far more effective in restoring habitats. Restoration of fourth order and larger alluvial valley streams, areas identified as having the greatest potential for fish production in the Pacific Northwest, will require natural watershed and riparian processes to be re-established over the long term. Reeves et al. (1991) described numerous structure and habitat manipulations (gravel cleaning, gabions, weirs, log sills, cedar baffles, fishways, boulders, log structures), and provided an evaluation of their use and applicability for variable life history requirements and differing watershed settings. They cautioned that much work has been done with very little pre- and post-evaluation of the results, and that successful future projects will depend upon careful evaluation of existing projects. Reeves et al. (1991) concluded that 1) habitat rehabilitation should not be viewed as a substitute for habitat protection, 2) prevention of initial habitat degradation is more economical of total resources than repairing that degradation, and **3)** some damage to streams is simply irreversible.

8.4 Forestry Practices

As linkages between upland and riparian forest management and aquatic ecosystems have become better understood, a gradual evolution toward more ecologically sound forest practices has occurred. Each of the States in the Pacific Northwest has adopted forest practice rules that provide greater protection to riparian areas along fish-bearing streams, and certain damaging practices (e.g., splash damming) have been eliminated altogether. In this section, we briefly review methods for minimizing the effects of forest practices on aquatic ecosystems. More information is provided in this section than in those for other land uses both because of the substantial overlap between forested lands and salmonid distributions and because the literature relating forest practices to salmonid habitats is far more extensive.

The effects of forest practices on watershed processes and aquatic ecosystems are influenced by the harvest schedule, methods, equipment, and unit location as well as by the site preparation methods, intermediate treatments, and the road location, construction, and maintenance. To minimize impacts, practices can be selected that are least disruptive to natural watershed processes. Forest practices that will most effectively protect stream ecosystems vary with local physical and biological characteristics.

8.4.1 Upland Forest Management

Although riparian activities pose the greatest risk to salmonids, upland practices affect surface erosion, mass wasting, hydrologic processes, and nutrient dynamics; therefore, these must also be considered.

Silvicultural Systems

Rotation schedule influences watershed dynamics by determining the frequency of disturbance to the watershed, the total area disturbed, and the quantity of materials delivered to a water body through time. Most commercial forests in western Washington, Oregon, and California have been harvested on a rotation of 45–100 years (Frissell 1991; Hicks et al. 1991a), though shorter rotations are becoming more common. Harvest rotations are typically longer for forests east of the Cascade Crest because climatic conditions lead to slower growth rates of trees.

Effects of harvest on stream temperatures, hydrology, surface erosion, and the probability of mass failures are generally greatest in the years immediately following logging because the degree of devegetation and soil disturbance is highest during this period. Lengthening the harvest rotation decreases the time that the landscape is disturbed and reduces the probability of catastrophic events. For example, if the risk of landslides is increased for a period of 15 years after logging, then a stand managed on a 60-year rotation will be vulnerable for 25 years per century versus 15 years for a stand managed on a 100-year rotation (Frissell 1991). On a regional scale, if the average rotation is 60 years then 25% of the landscape is vulnerable to landslides at any time, versus 15% with a 100-year rotation or 5% with a 300-year rotation.

Harvest can generally be divided into "even-aged" and "uneven-aged" methods (Young and Giese 1990). Even-aged methods are those in which the timber stand consists of trees of similar age and size and include such methods as clear-cutting, seed tree methods, and shelterwood cuts. Uneven-aged methods consist of those where trees are selectively harvested and where the resulting stand consists of trees of varied ages.

Clear-cutting has been the dominant harvest method in forests of the Pacific Northwest since the turn of the century (FEMAT 1993). Clear-cutting is potentially more disruptive of natural watershed processes—including hydrology, sediment transport, energy transfer, nutrient cycling, and stream habitat development—than other methods because virtually all vegetation is removed and soil is usually highly disturbed.

Patch cuts are relatively small clear-cuts distributed over the landscape. The effectiveness of patch cutting in mitigating effects on watershed processes depends on the size and location of the

harvest units and the percentage of the watershed harvested. Small patches may reduce hydrologic impacts because vegetation in areas surrounding the cut may take up some of the additional available water. However, compared with a single large clear-cut, multiple patch cuts may facilitate greater snow deposition, resulting in higher runoff during rain-on-snow events. While several patch cuts may be less disruptive to soil and hydrologic processes than a single clear-cut of equivalent total area, the resulting fragmentation of the watershed may be more detrimental. Dispersing timber harvest units and roads increases the probability of multiple chronic landslides across the landscape, rather than concentrating disturbances in particular watersheds. Also, for some wildlife species, large, intact systems function better as habitats than highly fragmented systems. Furthermore, multiple patch cuts may require more road mileage than a single harvested area, further diminishing the ability of habitats to support certain wildlife species, These tradeoffs should be considered when developing timber harvest strategies.

Seed tree and shelterwood cuts differ from clear-cuts in that some trees are left on site to provide seed sources for regeneration and, in the case of shelterwood cuts, to provide some shade for seedlings. Following the establishment of seedlings, the remaining large trees are removed, leaving an even-aged stand. Both of these methods can potentially reduce hydrologic effects and surface erosion. Trees that remain on site may exhibit compensatory growth in response to increased water availability, thereby minimizing increases in runoff. Standing vegetation also serves to reduce erosion, although seed trees are likely to have minimal effect on the probability of mass failures.

Selective harvest methods entail the removal of only a portion of the merchantable trees from a cut. Traditionally, only the largest trees were removed. Alternatively, harvest may involve selective removal of younger trees while leaving large, older trees standing. New forestry or "structural retention" techniques (sensu Swanson and Berg 1991) involve selecting a set of windfirm dominant and codominant trees to be retained for a rotation or more, thereby restoring large-diameter trees that are typically absent from most industrially managed forest lands. New forestry practices may also include retention of snags and downed wood, maintaining all size and age-classes of trees, and leaving patches as refugia. The intent of all of these techniques is to hasten the development of the characteristics and diversity typical of old-growth and late-successional forests that are desirable to some forms of wildlife, including the northern spotted owl and marbled murrelet. However, it is not known whether such measures will benefit other late-successional forest species (FEMAT 1993). The benefits of selective harvest vary depending upon the percentage of the basal area removed and the composition of the remaining stand. If properly done, selective harvest can maintain stream shading and input of allochthonous materials, minimize disturbance to soils, reduce soil compaction by ground-based equipment, and minimize effect on hydrologic processes. Disadvantages of selective harvest may include increased frequency of disturbance, suboptimal regeneration of trees, increased density of roads, and increases in harvest costs under some circumstances.

Harvest (Yarding) Systems

Considerable disturbance to soils occurs during skidding or yarding of logs from the stumps to landings. Several systems have been devised for yarding that differ in their impact to soils. Ground-based operations using tracked or rubber-tired skidders generally result in the greatest disturbance to soils both in terms of degree of compaction and the percentage of total area affected. Such disturbances increase surface runoff and erosion potential. Because most compaction occurs with the first few passes of heavy equipment, soil disturbance can be reduced by establishing designated skid trails, thereby minimizing the area receiving traffic. Because compaction is highest when soils are moist, impacts may be further reduced by skidding either during the dry season or when the ground is frozen.

In cable-yarding systems, logs are attached to a cable and dragged to the landing. In general, a lower percentage of the logged area is disturbed with this method; however, skid marks may channel water, thereby facilitating erosion. High lead systems are similar, with the cable running through an elevated pulley. Skyline systems lift part of the log off the ground as it is dragged to the landing site, further reducing the impact to soils. Full suspension methods lift logs completely off the ground. Helicopter logging has also been employed to reduce the need for roads and ground-based equipment. In general, the area affected by logging equipment and percentage of bare soil remaining is greatest for tractor and cable logging, intermediate for highlead systems, and least for full suspension, skyline, and helicopter logging (Pritchett and Fisher 1987).

Site Preparation

Regeneration of coniferous vegetation in some regions involves reducing shrubs and deciduous trees, eliminating logging debris, and preparing soils for planting. Three techniques have been used for site preparation: burning of slash, mechanical clearing of vegetation, and chemical treatments. As noted in

Section 3.3, the effects of fire on soil characteristics vary with the intensity of the burn. High-intensity fires can eliminate litter layers and create hydrophobic conditions in surface soils, thereby increasing the amount of exposed soil and the potential for surface runoff. Low-intensity burns, on the other hand, generally cause minimal damage to soils (Pritchett and Fisher 1987), and are therefore less likely to result in changes to hydrologic or erosional processes.

Mechanical clearing potentially has the greatest effect on soil conditions and hydrologic processes, and often causes greater damage to soils than the initial logging and yarding. Because heavy equipment must travel over most or all of a site, the potential for soil compaction and exposure of bare soils is high, which can lead to increased surface runoff and erosion (Pritchett and Fisher 1987). In addition, valuable topsoil may be redistributed, with much of it ending up in burn piles.

Chemical treatments may be least physically damaging to a logged site; however, care must be taken to ensure that chemicals do not reach stream systems. Limiting spray operations to calm days, using the minimum effective concentrations, and refraining from spraying in riparian zones can minimize the risk of exposure of aquatic organisms to toxic chemicals. In addition, applications can be timed so as not to overlap with sensitive life-history stages of fish or other aquatic biota. Norris et al. (1991) concluded that the herbicides used in forestry are relatively immobile in soils and that leaching into subsurface waters is less likely in forested soils than in other environments. From a hydrologic standpoint, application of herbicides is likely to extend the period of increased water yield from a site by reducing evapotranspiration losses. Conversely, at sites where deciduous vegetation will replace coniferous trees if untreated, chemical treatments may prevent reductions in summer streamflow that may occur over longer periods of time.

intermediate Treatments

Intermediate treatments are actions designed to enhance tree growth, and include thinning, pruning, and fertilization. The impacts of thinning and pruning activities are generally related to the type of equipment used and the care demonstrated by the operator. Lighter equipment will generally cause less disturbance to soils and ground cover than heavy equipment. In a well planned thinning, the use of equipment will be restricted to designated roads and skid trails, thereby minimizing additional soil compaction. Indiscriminate use of equipment, on the other hand, can result in compaction of soils that were left intact during the original harvest operation. Provided that additional soil compaction does not occur, thinning or pruning is unlikely to have a significant effect on water balance in a forest stand because remaining vegetation will take up additional water that becomes available. The effects of fertilization on aquatic systems can be minimized by refraining from fertilizing riparian zones. Fertilizing in ephemeral channels can lead to high concentrations of nitrogen in downstream areas when rainfall begins in the fall and fertilizers are mobilized (Norris et al. 1991). Thus, foresters should avoid applying fertilizers near all permanent streams, ephemeral streams, and drainage channels.

Road Construction and Maintenance

Logging roads are primary sources of sediments to streams, both through chronic erosion and as trigger points of mass failures. In addition, the higher the road density within a watershed, the greater the probability of significant alteration to hydrologic processes. Impacts can be substantially reduced by careful placement, construction, and maintenance of roads. Furniss et al. (1991) provided a thorough discussion of ways for minimizing the effects of road construction on aquatic ecosystems. Table 8-1 summarizes their recommendations, and much of the information in this section comes from their review. In addition, the forest practice rules of most western States contain guidelines for minimizing impacts of forest roads.

Minimizing total roaded area and careful placement of roads are the most fundamental means for reducing sediment inputs to channels. Long-range planning of road systems within a watershed helps minimize total surface area of roads and can reduce construction costs as well. In general, roads should be located away from stream channels, particularly in steep terrain where the likelihood of fill material washing into the stream is high. However, it may be preferable to construct roads in valley bottoms rather than on slopes that have a high probability of failure. In these instances, a buffer strip between the road and stream can reduce disturbances to the channel. Roads located too near the active channel in unconstrained reaches can impede the natural lateral migration of the stream channel across the floodplain, disrupting natural processes of erosion and deposition. Locating roads on ridgelines, as opposed to mid-slope areas, and on dry soils instead of in wet areas, also minimizes erosion risks (Furniss et al. 1991). Knowledge of local soils and geology is essential to ensure wise placement of roads. Hummocky ground, jack-straw trees, and sag ponds are often good indicators of unstable hillslopes subject to slumping or slides. Creek crossings are frequently the sites of significant erosion. Minimizing the number of crossings is both desirable and required under forest practice rules of some States

Table 8-1. Recommendations for minimizing impacts of forest roads on aquatic habitats. Recommendations based on Furniss et al. (1991).

Impact	Recommendation
Drainage	Disperse drainage rather than concentrating it.
	Avoid discharging large amounts of water into non-drainage areas.
	Avoid altering natural drainage patterns by means of water bars or culverts.
	Use outslope drainage to disperse runoff.
	Where inslope-and-ditch drainage is used, relieve the ditchline of drainage at frequent intervals.
	Use discharge pipes to route water away from fill slopes.
	Design drainage structures to accommodate peak streamflow based on at least 50-year-interval flood.
	Control scouring at culvert outlets with energy dissipators such as heavy rock, rip-rap, or other materials.
Stream crossings	Avoid channel width changes and protect stream banks with rip-rap or other retaining structures.
	Use retaining walls to reduce excavation near stream channels.
	Design road to approach creek crossings at right angles.
	Design crossings so they will not divert water down the road.
	Install instream culverts at angles and heights that allow fish passage. Culverts should be placed below original stream bed and gradient should be less than 1%.
Road beds, cut and fill slopes	Use minimum design standards for road width, radius, and gradient.
	Minimize excavation by using natural features.
	Design cut slopes to be as steep as practical.
	Where practical, surface roads to control erosion.
	Remove earth material and debris from streambanks to prevent them from being washed into the stream.
	Restrict gravel extraction to areas above high-water level of design flood.
	Do not incorporate organic materials into road fills.
	Use end hauling rather than side casting on steep slopes to minimize risk of fill-slope failures.
	Minimize height of cut slopes to reduce risk of failure.
Road location	Avoid mid-slope locations in favor of higher, flatter areas (ridgetops).
	Do not locate roads within inner valley gorge.
	Avoid slopes with excessive wetness.
	Avoid slopes requiring large cut and fill areas.
	Locate roads to minimize roadway drainage area and to avoid modifying the natural drainage areas of small streams.
	For valley-bottom roads, provide a buffer strip of natural vegetation between the road and stream.
	Locate roads to take advantage of natural log-landing areas.
	Minimize the number of stream crossings.
	Locate stream crossings to minimize channel changes.

(e.g., California). When crossings are unavoidable, they should be located where the amount of channel modification and fill material is minimized (Furniss al. 1991).

The principal considerations in designing and constructing roads to minimize effects on salmonid habitats are ensuring adequate drainage, preventing excessive sedimentation, and providing for fish passage at stream crossings. Sediment transport is generally tightly coupled with the routing of water on the landscape. Thus, most sound design and construction techniques are devoted to maintaining natural drainage patterns, preventing the concentration of runoff, avoiding discharge of water onto unstable fill slopes, and designing structures to accommodate extreme hydrologic events (Table 8-1).

Regular and timely maintenance of logging roads helps ensure that drainage and erosion control structures are functioning properly and allows identification of problems that could have adverse consequences. The costs associated with maintenance are generally low compared with reconstruction costs after a significant failure. Grading roads to ensure outsloping surfaces, and clearing of drainage ditches and culverts can ensure that drainage occurs as intended. Where problems are observed, installation of additional ditch-relief culverts or large culverts may alleviate erosion and drainage problems. Seasonal road closures may also be an effective way to reduce sediment delivery to streams.

Once harvesting has been completed at a site and the road is no longer needed, reseeding of the road bed with grasses reduces the amount of exposed soil and thereby decreases surface erosion. However, this practice has little effect on the potential for deep mass wasting. Recently, there has been growing support for revegetating or decommissioning roads by pulling sidecast material back onto the road bed and reforming the natural slope (Harr and Nichols 1993). Decommissioning of roads involves disturbance to restore natural morphology. Nevertheless, Harr and Nichols (1993) reported that decommissioned roads and landings sustained little or no damage following two significant rain-on-snow events that caused substantial damage to main haul roads in northwest Washington. Similarly, Weaver et al. (1987) reported that obliteration of problem road surfaces and fills, deconstruction of stream crossings, and re-contouring of disturbed slopes were effective techniques for reducing sediment input to streams in Redwood National Park. Other techniques aimed at surface and rill erosion problems were less successful at reducing sediment delivery and tended to be more costly. Both Harr and Nichols (1993) and Weaver et al. (1987) noted that a careful survey of road conditions that allowed them to identify significant problem areas was essential to successful cost-effective application of rehabilitation techniques.

8.4.2. Riparian Forest Management

Floodplain and riparian forests in the Pacific Northwest once supported some of the largest and fastest growing trees, and were among the first forests that were logged because of the relative ease of transporting logs via waterways. Recent research, however, has recognized the importance of floodplains, floodplain wetlands, and riparian zones for storing and slowing floodwaters, absorbing pollutants from runoff, reducing sediment delivery to streams, maintaining channel complexity, supplying shade and large woody debris, providing shallow water areas for feeding and spawning fish, and supporting a highly diverse community of plants and animals (BLM et al. 1994; Cederholm 1994). The practice of leaving riparian buffer strips along streams is now widely applied and is viewed as perhaps the most important aspect of protecting stream habitats from the effects of logging and other land-use activities (Cummins et al. 1994).

Three important considerations in establishing buffer zones are: 1) the width of the buffer zone, 2) the level of activity allowed within the riparian zone, and 3) whether riparian buffers are needed for tributary streams that do not contain salmonids. Appropriate buffer widths are the topic of much debate and a number of alternative approaches for determining adequate buffer widths have been proposed (FEMAT 1993; Cederholm 1994; FS and BLM 1994a; Cummins et al. 1994). The appropriate width of buffer zones depends on the specific functions that are being considered. Figure 3-2 illustrates generalized curves for the zones of influence of riparian vegetation relative to key riparian functions (FEMAT 1993). Litter inputs and bank stability are generally provided by trees within 0.5 potential tree heights of the channel. Shading and large woody debris are provided by trees farther from the stream channel; in some instances, significant amounts of large wood may be carried to the channel in landslides or debris flows originating outside of the riparian zone. The effect of vegetation on sediment and nutrient inputs may extend even farther from the channel, though these influences are more difficult to define. Complete protection of salmonid habitats requires that all of these functions be maintained. A more thorough discussion of riparian management practices, including State riparian protection rules, is presented in Part II, Section 14.2.3 of this document.

The influence of riparian vegetation also depends on physical and biological characteristics of the specific location, including topography, soil type, geology, and vegetative cover. For example, the

likelihood that large logs will end up in the stream channel is greater on steep slopes compared to gentle slopes. The effectiveness of riparian vegetation as a sediment filter depends on slope and soil type. Topographic shading may reduce the importance of shading by vegetation in some locations. The FEMAT approach to riparian buffers establishes buffer widths based on stream and land classifications; these widths can be adjusted if it is demonstrated, through watershed analysis, that riparian conservation objectives will not be compromised. This approach acknowledges that critical instream characteristics can be maintained with variable buffer widths determined in accordance with site conditions.

The second significant consideration in the management of riparian areas is the level or intensity of disturbance allowed within them. State forest practice rules generally allow harvesting of timber within riparian areas but at levels lower than in surrounding uplands, which are commonly clear-cut. Depending on the State, specific criteria establish the number of trees, species composition, basal area, overstory and understory canopy cover, or other measures of the vegetation to be left within the riparian zone (reviewed in Section 14.2.3). Deviations from standards (both more and less conservative) may also be granted based on site inspection or production potential. The implicit assumption of these rules is that some level of disturbance within the riparian zone is acceptable and will have minimal affect on salmonids and their habitats. For example, under Washington's forest practice rules, shade retention requirements for temperature control vary depending on whether he stream is classified as sensitive; thus, incremental temperature increases may be allowed that, while not causing direct physiological stress to aquatic organisms, may influence ecological interactions (see Chapter 4). **An** alternative view is that the target of riparian management should be no impairment of riparian function and that downstream and cumulative effects must be considered. Cederholm (1994) proposed that riparian zones should be identified and buffer zones should be established around the riparian zone to prevent modification of riparian function.

Where riparian zones have already been altered by human activity, the long-term prospects for recovery of large conifers may be limited without active manipulation of riparian vegetation. For example, many riparian zones in coastal forests have been converted to dense, alder-dominated stands that leave little opportunity for conifer regeneration. In these instance, thinnings can be used to create openings that allow sufficient light for conifer re-establishment. For such activities, leaving heavy equipment outside of the riparian zone can ensure that soils and streambanks are not disturbed.

Finally, impacts of logging can be reduced if buffer zones are left around small headwater streams that themselves do not support salmonids (Cumins et al. 1994). In particular, steep headwater drainages are frequently the trigger points of landslides. Minimizing road construction and logging around first order and temporary channels can prevent frequent mass soil movements that propagate downstream, to the detriment of salmonids.

8.5 Range Practices

Livestock grazing occurs on approximately 70% of both Federal and nonfederal lands in the West (GAO 1988b; Fleischner 1994), making it the most widespread land use in the region. Many wildlife refuges, wilderness areas, and even some national parks (e.g., Great Basin National Park) are grazed by domestic livestock. Since the 1930's, rangelands in the Pacific Northwest have benefited from less intensive grazing; however, the majority of western rangelands remain in deteriorated condition. Although thorough surveys of range condition on both private and public lands are lacking, the available evidence indicates that between 50% and 65% of rangelands are in poor or fair condition (Busby 1979; GAO 1991; Heady and Child 1994). An even higher percentage of western riparian areas are in degraded condition. On BLM and Forest Service lands in Colorado, Arizona, Nevada, and Oregon, between 60% and 93% of riparian areas are considered to be in poor or fair condition (GAO 1988a; Armour et al. 1994). Poor upland conditions increase sediment loads and alter hydrologic regimes, leading to channel incision, channel widening, and further deterioration of riparian zones. Similarly, damaged riparian areas are unable to buffer streams from changes brought on by degradation of upland areas. Thus, restoring salmonid habitats in rangelands requires improving livestock management in both upland and riparian areas.

8.5.1 Upland Range Management

Although strategies for improving salmonid habitats in rangeland streams have to date focused primarily on modifying grazing practices within the riparian zone, effects of grazing on hydrologic and sediment transport processes necessitate improvement of range practices in upland areas as well. Hydrologic changes occur in response to loss of vegetation or change in plant species composition, **as** well **as** to changes in soil permeability brought on by reduced organic content, splash erosion, and trampling by livestock. Similarly, sediment transport processes are linked to vegetation cover and the routing of water from the hillslope to the stream. Consequently, restoring the

natural rate of water and sediment delivery streams can be attained by restoring more natural vegetation assemblages in uplands.

The retention of adequate vegetative cover in uplands is critical to minimizing hydrologic and sediment impacts and can be achieved through a variety of means including 1) lowering livestock stocking rates; 2) controlling animal distribution through fencing, herding, salting, and watering; 3) changing species composition of livestock; and 4) altering the timing of grazing (Heady and Child 1994). Each of these practices can influence the percent of forage that is utilized, the composition of vegetation that remains on site after grazing has ended, and the degree of soil disturbance.

Reducing stocking rates not only provides greater protection to streams and watershed, it can result in improved condition and value of livestock as well. At high stocking rates livestock do not gain weight quickly or at all, or they lose weight and condition. At the same time, the range deteriorates or fails to recover, costing ranchers money over the long term. In addition, high stocking rates maximize financial losses when livestock prices fall between the time of calf acquisition and sale. Yield rises constantly with increased livestock density (assuming a constant environment) towards some optimum, then falls sharply slightly beyond that optimum production level. This makes it very difficult to select an optimum stocking rate in a predictable environment, let alone a highly variable one. Heady and Child (1994) report that, for both environmental and financial reasons, increasing numbers of managers are shifting toward lower stocking levels.

Livestock tend to concentrate in areas near water, shade, preferred vegetation, salt, and on relatively level topography. As a result, these heavily used areas may become overgrazed and trampled, leading to erosion and hydrologic disruption. Development of alternative water sources, salting, fencing, and herding can all be used to ensure more uniform utilization of forage (and hence remaining cover) and to reduce impacts associated with soil compaction.

Although common in eastern and southern Africa, where native diseases limit domestic livestock, game ranching and cropping have not been widely adopted in this country. Livestock grazing benefits wildlife species that prefer habitats altered by livestock, but harms those species preferring natural habitats, or those competing with livestock for food (Heady and Child 1994). Therefore, depending on the desired species of wildlife, livestock may need to be reduced or removed. Native wildlife populations can sustain high harvest levels; for example, some deer populations have been harvested at rates of 33%–50% annually for decades without detrimental effects (McCullough 1979; Heady and Child 1994). Various game management and harvest programs are also possible on these lands, depending on whether the objectives are trophies, meat, or wildlife viewing. Selling prices range from several thousand dollars for a trophy animal to a few dollars for wildlife viewing. However, wildlife viewing is a nonconsumptive, repeatable activity. Benefits in addition to range improvement include lower management costs, leaner meat supply, and higher biological integrity.

Because plant responses to defoliation differ depending upon the season of grazing, vegetation can be protected by restricting grazing during certain times of the year. Four general types of grazing plans have been utilized: continuous, repeated seasonal, deferred, and rest (Heady and Child 1994). As the name implies, continuous grazing entails grazing throughout the growing season and usually some part of the dormant period, thus the length of time varies with climate. Repeated seasonal grazing refers to annually grazing the same pasture during a specific season, similar to the patterns of migratory wildlife. Deferred grazing means no grazing is conducted until key plants have completed reproduction. Rest once referred to a year without grazing but has since been generalized to any specified period. Where range condition is less than excellent or in arid and semiarid regions, range recovery may require many years of deferment or rest, The success of each is a function of site characteristics, periodic monitoring, and low stocking densities. These will determine which pastures to graze, which season or year to graze them, and for how long.

8.5.2 Riparian Range Management

Western riparian areas are among the most productive ecosystems in North America, yet their present condition is believed to be the worst in American history, largely because of livestock grazing (Fleischner 1994). Acknowledging the need to manage the entire watershed, Barrett et al. (1993) established a BLM goal of 75% or more properly functioning riparian wetlands by 1997. They defined proper functioning condition as adequate vegetation, landform, or large woody debris to 1) dissipate stream energy associated with high flows, thereby reducing erosion and improving water quality; *2)* filter sediment, capture bedload, and aid floodplain development; 3) improve floodwater retention and groundwater recharge; 4) develop root masses that stabilize streambanks against cutting action; *5)* develop diverse ponding and channel characteristics to provide the habitat, water depth, flow duration, and stream temperature necessary for fish production, waterfowl breeding, and other uses; and 6) support greater biodiversity.

Heady and Child (1994), Platts (1991), Chaney et al. (1990), and Kauffman and Krueger (1984) identified numerous options to be considered singly or in combination for achieving grazing goals while maintaining or improving fish habitat. They include resting from grazing, controlling livestock distribution, controlling livestock numbers, controlling forage use, controlling timing of forage use, grazing the type of livestock best suited for a given area, and artificially rehabilitating stream riparian ecosystems. Elmore (1992) and Platts (1991) provide greater elaboration on specific grazing strategies and their influence on riparian ecosystems and fish habitats (Table 8-2).

Grazing stress and trampling impacts are a function of how and when a given strategy is used and not simply on the total number of animals grazing at a particular site. Strategies that include corridor fencing, rest rotation with seasonal preference, and complete rest from grazing provide the greatest potential for rehabilitating degraded riparian areas (Platts 1991). Other grazing strategies may be effective under specific circumstances, depending on local climate, hydrologic conditions, soils, stream character, and plant species composition (Chaney et al. 1990; Elmore 1992). Selective timing for rotation grazing and strategies that allow growth of riparian vegetation during critical periods may be just as effective as reducing grazing intensity in some areas (Elmore 1992). Seasonal grazing strategies can also minimize trampling impacts to streambanks (Kauffman and Krueger 1984; Chaney et al. 1990). Having a clear set of riparian objectives and understanding vegetation potential at the site are critical to identifying effective riparian grazing strategies. It is also important to note that rangeland streams have unique attributes that may make them more vulnerable to anthropogenic stresses. For example, rangeland streams typically experience low or highly fluctuating flows and rocky or highly alkaline soils that severely limit riparian vegetation (Crouse and Kindschy 1981). If other preexisting stress levels are high, the capacity to absorb additional grazing stresses or to recover when they are removed may be low. Thus, effective grazing strategies must integrate the natural potential and the expected grazing stress for a given stream reach (Elmore 1992). In the absence of site specific information, deferment of grazing until riparian vegetation has returned to pre-grazed conditions remains the safest course of action.

Numerous case histories document the effectiveness of various grazing strategies in improving riparian conditions. The GAO (1988b) and Chaney et al. (1990) recently reviewed riparian restoration efforts on BLM and Forest Service lands and reported substantial improvement in riparian and stream conditions in many instances. Although riparian fencing and reductions in stocking rates generally proved to be most effective measures, deferred grazing, rest-rotation, daily herding, and development of off-stream water sources also yielded positive results. Ecological benefits noted included improvements in riparian grasses and shrubs, regeneration of cottonwood trees, increases in summer streamflow, reductions in summer water temperatures, narrowing and deepening of stream channels, increased pool area, increased stability of streambanks, improved substrates, and increased fish density.

In their review, Kauffman and Krueger (1984) cited studies indicating that one-to-two years of rest out of three provided improved riparian vegetation as long as forage consumption was below 60%–65%.They also noted that although rest-rotation strategies may improve condition of vegetation, increases in trail formation and trampling may cause streambank erosion and instability. Finally, they note that daily herding of sheep from stream bottoms improves utilization of upland forage while providing greater protection to riparian areas.

As in uplands, improvements in grazing practices within riparian areas can provide economic benefits to ranchers. The GAO (1988a) report cites several instances where animal unit months increased following recovery of riparian vegetation. Livestock excluded from riparian zones were forced to graze on underutilized upland vegetation, resulting in healthier, heavier livestock.

Frequently, riparian restoration efforts on rangelands have involved both changes in grazing practices and placement of instream structures. Kauffman and Krueger (1984) reported that better livestock management was a less costly strategy than instream structures for restoring channel integrity. They added that the need for instream structures is negated by rest from grazing and that structures are often ineffective when not accompanied by modification or cessation of grazing within the riparian zone. Similar conclusions were reached by GAO (1988a), Chaney et al. (1990), and Beschta et al. (1991). Instream structures were considered by Heady and Child (1994) to treat the symptoms of improper grazing rather than the causes. Thus, instream structures are likely unnecessary for rangeland stream restoration except under unusual circumstances, and they should not be used as a substitute for improved management of livestock.

8.6 Agricultural Practices

Agriculture and human settlement began on floodplains because of the availability of water and the fertility of soil. As a consequence, floodplains

Table 8-2. Evaluation of the effects of various grazing strategies on riparian habitats. From Platts (1991). Used with permission from the publisher.

Strategy*	Level of vegetation use in riparian area	Control of animal distribution	Stream-bank stability	Brushy species condition	Seasonal plant regrowth	Riparian rehabili-tation potential	Rating†
Continuous season-long use (C)	Heavy	Poor	Poor	Poor	Poor	Poor	1
Holding (S or C)	Heavy	Excellent	Poor	Poor	Fair	Poor	1
Short-duration, high intensity (C)	Heavy	Excellent	Poor	Poor	Poor	Poor	1
Three-herd, four pasture (C)	Heavy to moderate	Good	Poor	Poor	Poor	Poor	2
Holistic (C or S)	Heavy to light	Good	Poor to good	Poor	Good	Poor to excellent	2 – 9
Deferred (C)	Heavy to moderate	Fair	Poor	Poor	Fair	Fair	3
Seasonal suitability (C)	Heavy	Good	Poor	Poor	Fair	Fair	3
Deferred rotation (C)	Heavy to moderate	Good	Fair	Fair	Fair	Fair	4
Stuttered deferred rotation (C)	Heavy to moderate	Good	Fair	Fair	Fair	Fair	4
Winter (S or C)	Heavy to moderate	Fair	Good	Fair	Fair to good	Good	5
Rest rotation (C)	Heavy to moderate	Good	Fair to good	Fair	Fair to good	Fair	5
Double rest rotation (C)	Moderate	Good	Good	Fair	Good	Good	6
Seasonal riparian preference (C or S)	Moderate to light	Good	Good	Good	Fair	Fair	6
Riparian pasture (C or S)	As prescribed	Good	Good	Good	Good	Good	a
Corridor fencing (C)	None	Excellent	Good to excellent	Excellent	Good to excellent	Excellent	9
Rest rotation with seasonal preference (S)	Light	Good	Good to excellent	Good to excellent	Good	Excellent	9
Rest or closure (C or S)	None	Excellent	Excellent	Excellent	Excellent	Excellent	10

* C = cattle, S = sheep.

† Strategies are rated on a scale of 1 (poorly compatiblewith fishery needs) to 10 (highly compatible).

and riparian areas in agricultural lands remain among the most disturbed areas in the landscape, particularly because agriculture most often involves complete replacement of natural vegetation and repeated disturbance of soils through tillage. Regulations, management practices, and any other activities that completely protect floodplains, riparian areas, and uplands in a natural state ensure that channel and riparian functions are unimpaired. Recognizing that totally protecting large numbers of agricultural watersheds or floodplains is impractical under current demands for food and other crops, we focus in this section on management strategies that preserve most critical functions while allowing continued use of agricultural lands. These strategies are aimed at conserving water and soil, protecting and restoring riparian vegetation, and minimizing use of chemical pesticides and fertilizers.

8.6.1 Upland Cropland Management

Current agricultural practices offer ample opportunities for conservation. Of all the water diverted and consumed in the Pacific Northwest, approximately 90% is used to irrigate crops (Wilkinson 1992), and this percentage is even higher in several sub-basins east of the Cascade Crest (Muckleston 1993). Irrigation and associated water quality problems are the major stressor in reduced salmon runs in the Yakima, Walla Walla, and Umatilla Rivers (NPPC 1986). Where irrigation withdrawals are substantial, one of the most important current management actions for restoring salmonids is maintaining adequate year-round instream flows. Without sufficient flows, other restoration activities are likely to be ineffective. Instream flows can be maintained through a combination of instream water rights and water conservation efforts. State and Federal fish and wildlife agencies have established most water rights, but landowners in some States, including Oregon, have also granted or sold water rights to fishermen's groups or the State, generating more net income than they did from marginal crops (Wilkinson 1992). One of the potentially most useful tools for maintaining or restoring instream water is the 1908 U.S. Supreme Court decision on *Winters vs. United States*, which decreed that Indian tribes possessed water rights that were superior to those established by State law. Another legal approach involves the common-law public trust doctrine, which holds that the rights to water on larger water bodies cannot be controlled by a single **part** of the population (Wilkinson 1992) and that private property owners are prohibited from acting in a manner inconsistent with public trust interests (Johnson and Paschal 1995).

Water conservation is the most fundamental way to provide more water for aquatic life. Environmental concerns surrounding the construction of new dams on salmonid-bearing streams underscore the need to improve the efficiency with which water is used in agriculture. Water use can be decreased by timing irrigation to coincide with periods (both daily and seasonal) of low solar radiation to minimize evapotranspiration losses. Installation of cement-lined canals can reduce transpiration losses by noncrop plants that typically grow in unlined irrigation ditches. Use of drip or trickle irrigation systems instead of flood irrigation minimizes evaporation losses as well as reducing the need for weed control. Conversion of water-intensive crops such as rice and alfalfa to more drought-resistant crops can also minimize water use. Leveling of fields with laser technology has been employed to minimize runoff of irrigation water from croplands. Each of these methods maximize the efficiency with which water is used while simultaneously reducing chemical and thermal pollution associated with irrigation return flows. Effective regulatory tools for water conservation include monitoring and taxing water use, and using graduated pricing for the water consumed (Wilkinson 1992).

Screening of irrigation canals and pump intakes is also essential to protect salmonids. An investigation of 225 intakes along the Columbia River during the late 1970s indicated that 70% lacked proper screening (Swan et al. 1980); subsequent surveys concluded that 30% of intakes remained improperly screened after irrigators were notified of inadequacies (Swan et al. 1981). In many instances, existing screens may be sufficient to protect outmigrating smolts but insufficient to prevent entrainment of smaller fry (Palmisano et al. 1993a).

Control of sedimentation from agricultural lands remains a significant concern in many lowland streams in the Pacific Northwest. Nationwide, five of the six most popular soil conservation programs funded by the Agricultural Stabilization and Conservation Service in recent decades involved increased vegetative cover (ASCS 1992). In southeast Washington, replacement of row crops and small grains with permanent vegetation or hay and pasture was predicted to reduce erosion rates to 0.1%–0.01% of their former level. In the same study, various forms of conservation tillage reduced soil erosion by 13%–95%, depending on precipitation (SCS et al. 1984). Grassed waterways continue to be popular as a means to limit soil erosion and many enlightened farmers leave riparian buffers along surface waters. Dairy farms typically have wet weather controls for limiting run-off from manure heaps and cattle are fenced from stream access. Currie (1994) recommended greater use of existing incentives and disincentives. Among incentives, he included greater tax abatements through the Washington Open Space

Program, and higher Federal subsidies for implementing best management plans. At the same time, Currie proposed that farms not implementing best management plans be subjected to reduced Open Space exemptions and higher fines for farm pollution. Federal laws also encourage farmers to take highly erodible lands out of production by making farmers that cultivate such lands ineligible for Federal price supports, crop insurance, loans, or disaster payments (see Chapter 9). Watershed analysis, including risk assessment, management plans, and monitoring, is also proposed as a successful tool that could be adapted from forestland management to farmlands (Currie 1994).

Reducing the use of chemical pesticides and fertilizers is another important conservation strategy for agricultural lands. Organic farming and integrated pest management are also growing in popularity on small agricultural operations. Demand for biocide-free crops continues to rise along with concerns about the human health and ecological effects of pesticides. In addition, the cost of biocides and problems with neighboring landowners and contaminated ground water has restricted their use in some areas. These changes have mostly occurred on small farms that can more effectively implement integrated pest management. Where chemicals are used, contamination of streams can be minimized by applying chemicals at their minimum effective concentration, by spraying during periods of low wind, and by maintaining no-spray riparian buffers.

8.6.2 Riparian Cropland Management

As in forest and rangeland management, the practice of leaving riparian buffer strips is central to conservation of streams and rivers in agricultural lands. Vegetated buffer strips greatly reduce the delivery of sediment and chemical pollutants from croplands. In addition, riparian buffers stabilize streambanks, provide shade, and contribute large wood to streams that frequently lack these attributes. Riparian forests, together with fencerows, frequently constitute important wildlife habitats in agricultural landscapes otherwise devoid of suitable habitats.

Also important to the restoration of streams and rivers in agricultural lands is the re-establishment of natural floods. Just as it is now acknowledged that large woody debris is critical for maintaining channel complexity, it is also apparent that floods are necessary to prevent channels from incising, redistribute coarse sediments, build floodplains, introduce large wood, and propagate natural riparian vegetation. Traditional State and local floodplain zoning and easements allow some measure of control over the type of activities that can occur on floodplains. This approach may be quite successful if those allowed activities cause only minimal disruption of the floodplain ecosystem (e.g., natural parks, fishing access points). Typically, however, floodplain zoning restricts only those activities that incur extensive damage during floods (e.g., structures), but allows other activities (e.g., logging, grazing, farming) that significantly change the characteristics of floodplain ecosystems and, hence, the functions they perform (Kusler 1979).

An example of a more protective approach is the Banner Drainage and Levee District in Illinois, which is being restored to lakes and wetlands (NRC 1992). Another approach is the Willamette River greenway in western Oregon that includes 255 river miles and includes sloughs and side channels (NRC 1992); however, agricultural lands are exempt from greenway regulations and in many reaches there is no natural floodplain or only a narrow strip of native vegetation. Oregon also has a program to provide tax relief to landowners that maintain natural riparian zones. Federal laws have sought to protect wetlands in agricultural areas by eliminating U. S. Department of Agriculture benefits (e.g., price and income supports) for areas where farmers cleared and drained wetlands for crop production (see Section 9.4). Water Quality 2000 (1992) and NRC (1992) both stressed the importance of protecting and restoring existing wetlands rather than trying to recreate them after they have been converted. Recognizing the ecological need for riparian and wetland areas to flood and the substantial cost to humans when they do, it may be more prudent to relocate activities from floodplains than to subsidize continued development of floodplains through channel maintenance, dam and levee construction, Federal flood insurance, and "disaster" relief (NRC 1992). The benefits include saving money, allowing natural processes to re-establish habitat, and reducing hazards to human residents.

8.7 Mining Practices

In 1872, hard rock mining was encouraged through legislation by the Federal government so that miners could easily obtain mining claims, produce metals, and settle the West. Increasing concerns for the ecological costs of mining, particularly pollution of streams and rivers by mining wastes, has prompted calls for more strict regulation of mining activities. Wilkinson (1992) proposed several policy options including banning patenting of Federal lands (obtaining ownership by simply paying $2.50–$5.00 per acre and investing $100 per year), setting strict reclamation requirements (including liability bonds), charging value-based royalties on minerals removed, and prohibiting mining in sensitive areas. He also recommended leasing, rather than selling, the land to miners and evaluating whether there is a net public benefit of the mining. The Western Division of the

American Fisheries Society (WDAFS 1994), in calling for reform of the 1872 mining law, recommended these same measures, as well as public participation in all aspects of mining regulation, programs for monitoring compliance, time limits on Plan of Operation approvals, and other environmental safeguards. Nelson et al. (1991) added that riparian and stream enhancement should also be part of the reclamation process.

8.7.7 Upland Mining Practices

Reclamation of mining sites typically focuses on preventing mine-generated solid wastes and toxic materials from entering waters. This can be most effectively achieved by restoring natural landscape contours, followed by re-establishment of vegetation. To allow restoration of natural vegetation on mined lands, it is critical that topsoil be set aside before mining begins (Nelson et al. 1991). Toxic materials should be buried below the root zone to prevent uptake of by plants. In addition, toxic wastes should be buried away from areas where leachates are likely to enter streams or groundwater. Ground water does not pass through these materials to streams or the water table. When the area is returned to its natural contours (this can be problematic when the volume of spoils exceeds that of the original ore), the soil is replaced and revegetated with the original flora or acceptable substitutes. Revegetation may require seeding or introduction of vegetative propagules, as well as tilling, mulching and fertilization. It is critical that lands be stabilized as soon as possible to limit erosion. Mining-generated solids and seepage or runoff from mines should be kept from streams by proper planning and control structures such as erosion barriers and lined ponds.

8.7.2 Riparian and Instream Mining Practices

Mining in or near streams requires additional precautions to those for uplands (Nelson et al. 1991). Effluents may be treated with hydrated lime or sulfite and then aerated to raise the pH and allow the metals to precipitate. Reverse osmosis and electrochemical precipitation are also effective following acid neutralization. If the channel form and substrate have been altered, a channel and riparian zone should be developed that allows normal ecological processes to occur. Nelson et al. (1991) stressed that such channels are not static; they should resemble the pre-existing channel in their bed, banks, riparian vegetation, and flows. Re-establishment of riparian woody plants may require transplanting.

Instream and floodplain aggregate mining poses special problems since, by its nature, this activity involves disturbance to channel morphology. In reviewing effects of gravel extraction on streams in Oregon, OWRRI (1995) made several recommendations for minimizing degradation to salmonid habitats: 1) prohibit, regulate, or otherwise manage small operations (i.e., less than 50 cubic yards); 2) conduct gravel removal in streams in a manner to minimize potential impacts on salmonid habitats; 3) allow gravel removal by bar skimming only under restricted conditions (i.e., where the gravel bar is not an active spawning, rearing, or feeding are for salmonids; where adequate gravel recruitment exists to replenish the bar; where berms and buffers can be used to control streamflow away from the excavation site; where gravel can be removed from above the low-water level during low flows; and where the final grading of the bar does not significantly alter the flow characteristics of the river at high flows); 4) restrict deep-water dredging for gravel production to areas where it is presently practiced; 5) do not allow a net loss of wetlands for all fill/removal operations; and 6) use biological streambank stabilization methods where possible. Secondly, they stressed the need for monitoring and research to evaluate impacts, improved database capabilities and use for the managing agency, implementation of GIS-based resource management, and allocation of sufficient funds to monitor resource abundance, conditions, and use. From a policy standpoint, they suggested that 1) the burden of proof of "no significant impact" should be shifted to permit applicants where proposed activities are expected to result in significant direct or indirect impacts to salmonids; 2) gravel extraction should not be allowed in reaches of ODSL-managed streams that support sensitive, threatened or endangered species; and 3) gravel extraction should not be allowed from reaches of ODSL-managed streams that are part of Aquatic Diversity Areas or that support source salmon populations. Finally, they concluded that gravel removal operations may provide potential opportunities for increasing salmonid habitats through reconnecting former floodplain gravel pits to riverine systems; using gravel mining as a potential method for creating wetlands, off-stream channels, lakes and ponds, or salmonid spawning beds; and using gravel mining to improve spawning areas by improving sediment quality, increasing channel sinuosity in streams that have been channelized or otherwise simplified. In all of these instances, active restoration should be ecologically based and carefully studied prior to implementation.

8.8 Urban Practices

Restoring and protecting salmonid habitats in urban areas is one of the most difficult challenges facing land managers because many disturbances to the urban landscape are essentially irreversible, barring a radical change in social values.

Urbanization fundamentally alters water quality in streams (Smart et al. 1985). The high percentage of impervious surfaces leads to increased runoff, making stream hydrographs much flashier. For example, a 20% increase in impermeable surfaces can double runoff in a storm event (Birch et al. 1992). Channel morphology is modified by intentional conversion of natural drainage channels into conduits and as a consequence of decreased channel stability resulting from higher peak flows. Riparian vegetation is also extensively modified, with gallery forests and shrubs being converted into buildings, roads, parking lots, and lawns. Protecting and restoring lakes and streams from the effects of urbanization, therefore, involve reducing the areal extent of urbanization, removing pollutants from the waste stream, and conserving natural channels (Wanielista 1978).

Increasingly, land-use planning is used to restrict urban development from most sensitive areas, although the effects of unplanned (with respect to aquatic resource concerns) development persist. Relatively successful examples of such planning include the California Coastal Commission, the Oregon Land Conservation and Development Commission, and the Tahoe Regional Planning Agency (Wilkinson 1992). The impact of urbanization also can be reduced by favoring high-density housing, by greater utilization of bicycles and mass transit, and by placing major transportation networks underground: in other words by designing cities for people instead of automobiles (Doxiadis 1971). Not only do these more rational city designs decrease the amount of impervious surfaces, but also they decrease the amount of pollutants collected by and discharged from roads. As with other land uses, rigorous basin planning is incorporated in successful urban planning (Birch et al. 1992).

The major way of removing urban pollutants from the waste stream is by industrial and sewage treatment plants. Most U.S. cities now have secondary sewage treatment, and many industries have tertiary or secondary treatment. Secondary treatment, however, only transforms wastes into nutrients that are then discharged into rivers. As water purification and waste treatment costs rise and growing human populations increase the demand for limited water, more households and municipalities begin water rationing and recycling. Household rationing has taken the form of xeriscaping in place of lawns, limiting lawn irrigation, smaller toilet tanks, low-flush toilets, composting toilets, rinse-only showers, and low-discharge shower nozzles (Wilkinson 1992). The major opportunity for domestic recycling is in the reuse of washwater on gardens and lawns and in toilets (Wagner 1971). Municipal rationing involves higher and progressive water and sewer rates as well as lawn-watering restrictions. Cities also recycle sewage water for irrigation or find it less expensive to purify chemically treated effluent for reuse than to withdraw lower quality water from rivers. Both approaches are in use elsewhere in this country (Wagner 1971) and in Europe. Industrial rationing and recycling have become more common as discharge permits became more restrictive.

Another substantial source of polluted waters is stormwater runoff from lawns, roofs, parking lots, and streets. These sources can be reduced by decreasing their surface areas as discussed above and by requiring point-source discharge permits on outfalls. More typical practices can be broken into nonstructural and structural approaches (Wanielista 1978). Nonstructural approaches include street cleaning (especially mechanical broom and vacuum sweeping), cleaning of catch basins, dust control, restrictions on the use of lawn chemicals, erosion control at construction sites, and the use of wetland systems (vegetated floodplains, marshes, ponds riparian zones) as natural filters. Structural management practices for reducing stormwater runoff problems include retention basins, constructed wetlands, land injection, rooftop and parking lot storage, and sediment traps. Illicit connections to storm drains are very common in cities (Birch et al. 1992). Improper connections can be located by associating chemicals in the effluent with likely producers, dye studies, and TV inspections. Birch et al. (1992) provide a thorough set of guidelines for controlling erosion and sedimentation from construction sites, including matting and mulching open soil, erosion barriers, sediment traps, interceptors and drains on cut-fill slopes, and removal of sediment from roads. Monitoring and maintaining control structures at such sites, especially during storms, is important.

8.9 Regional Planning and Management Efforts

Most management efforts for protecting and restoring salmonid populations have focused on the fish (harvest restrictions, fish passage, hatchery supplementation) or on aquatic habitats (water quality criteria, physical habitat structure, flow) within a limited area dictated by the particular land- or water-use activity. Although population- and site-specific efforts are an essential component of salmonid restoration, many issues related to long-term persistence of salmonids involve larger spatial scales and hence require statewide or multistate planning.

FEMAT (1993), PACFISH (FS and BLM 1994a), and INFISH (FS 1995) are examples of coordinated, Federal, land-use planning that, despite what some

perceive to be compromises and limitations, represents significant progress towards regional conservation. Several States have begun similar efforts. For example, Oregon's statewide land-use planning law is designed to protect forest, agricultural, and coastal lands from urbanization. Goal 5 of that law requires conservation and protection of lands needed for fish and wildlife habitats, water areas, wetlands, watersheds, and groundwater. Although it is statewide in scope, it is implemented and monitored at the county level with little statewide assessment of status or trends by the Department of Land Conservation and Development. Oregon's riparian set aside law, overseen by the Department of Fish and Wildlife, offers landowners tax deductions for protecting such areas. In response to Senate Bill 1125, Oregon Department of Forestry (ODF 1994) recently developed rules providing increased riparian protection for all fish-bearing forest streams. Levels of protection vary with water body use, type, and size. A higher design and maintenance standard for new stream-crossing structures was also promulgated. Aquatic diversity areas (Henjum et al. 1994), similar to FEMAT's key watersheds, have been mapped by the Oregon Chapter of the American Fisheries Society for the entire State. These areas, together with locations of unusually high salmonid production, have been incorporated into a framework for allocating salmon restoration funds (Bradbury et al. 1995). Protection and restoration of such areas throughout the region are necessary to preserve and expand salmonid populations that can support sustainable harvests.

The other States in the region have developed similar planning and management systems. For example, Washington Department of Natural Resources has rules for riparian protection that vary with water body use, type, and size. In addition, Washington promotes watershed analysis as a means of identifying sensitive and high-risk areas within watersheds, or to minimize disturbances to aquatic ecosystems resulting from forest practices. California's Coastal Zone Management Act restricts development on sensitive coastal and estuarine areas. All four States in the region have water quality (temperature and dissolved oxygen) standards for the protection of salmonids. Generally these standards would be protective if monitored and enforced. In addition, the States have local zoning laws restricting building types and densities, The water quality and land-use standards, however, differ from State to State and lack a statewide planning and monitoring design, let alone a regional one.

8.10 Individual and Social Practices

Direct alteration of habitat by humans remains the single greatest threat to both terrestrial and aquatic biodiversity (Noss 1992). Most habitat alterations affecting salmonids relate to resource consumption of some sort—the use of water, electricity, wood and wood products, meat and wool, food and nonfood crops, and mineral resources. Per capita consumption of resources is an order of magnitude greater in the United States than it is in much of the world. Therefore, each of us can minimize our indirect effects on salmonids by markedly reducing consumption of all resources. There are a number of things that we can do individually and as a society to begin these changes'and reduce our environmental impact.

8.10.1 Short-term Individual and Governmental Actions

The amount of water available for aquatic life in streams and lakes can be increased by reducing the amount diverted for domestic, industrial, and agricultural uses. Water conservation begins at home in how we shower, launder, flush toilets, landscape, irrigate, and use electricity. Those same functions of cleaning, cooling, waste disposal, irrigation, and power consumption offer opportunities for water conservation in industry and agriculture as well. Potential for increased efficiency in these water uses has been demonstrated by various voluntary and mandatory water conservation measures implemented during recent droughts in California, Oregon, and Washington, Key aspects in conserving water and electricity are accurate monitoring of uses, internalizing environmental costs associated with water use (e.g., dam impacts, hatchery operations, wastewater treatment), and progressive pricing so that greater use results in proportionately higher rates.

Wiser use and conservation of metals, particularly aluminum and heavy metals, would also reduce the demand for hydropower and hence the adverse effects of dams on anadromous salmonids. Over 40% of the aluminum used in this country is produced in the Pacific Northwest, and fully 20% of the total energy sold by BPA is used by aluminum smelters and other energy-intensive industrial processes. Excessive packaging results in enormous waste of aluminum. Every three months, Americans discard enough aluminum to rebuild the Nation's entire commercial air fleet. Recycling aluminum requires approximately 5% as much energy as refining the metal from bauxite, Because aluminum is an important component of many car parts, demand for aluminum can also be curtailed through reduced auto use and ownership, as well as greater reliance on mass transit and other forms of transportation.

As with aluminum, wood products have considerable conservation potential. Worldwide, humans used over 30% more wood per person in

1991 than in 1950, mostly as fuel, but in the developed countries per capita wood consumption has been declining for most of the century (Durning 1994). Other trends are less encouraging. The United States produces 26% of the world's industrial wood with Russia a distant second. Average house size in the United States has increased from 100 m^2 in 1949 to 185 m^2 in 1993. This represents 50%–100% more space per person than West Europeans and Japanese, respectively. World paper consumption has increased 20-fold since 1913, mostly in the highly industrialized nations. In 1960, the average family in the United States spent $500 per year on packaging, and the Nation as a whole paid $190 million for junk mail (Packard 1960). Currently, over half a million trees are used each week to print this Nation's Sunday newspapers, much of which consist of advertisements that many readers discard and that promote consumption of unneeded products. Demand for wood products can be reduced by creating "paperless" offices, decreasing packaging, recycling paper, and developing alternative sources of fiber. Recent development of chipboard has increased fiber supply options to include wood waste, previously undesirable weed trees, and agricultural wastes such as straw and hemp. Agricultural waste fibers and weed trees also hold promise for paper manufacturing. Current methods to reduce waste and increase recycling and manufacturing efficiency could halve present wood consumption in the United States (Postel 1994).

Because livestock production and commercial fish harvest have substantial effects on salmonids, it is useful to examine ways to reduce consumption of beef and fish. Alternative protein sources, such as grains and legumes, would reduce the demand for salmon harvest and the need for range-fed livestock. Only 10% of the protein ingested by cattle is converted to tissue; consequently, beef is a relatively inefficient source of protein for humans compared with grains and legumes. The growing number of vegetarians (currently estimated as 4%) in the United States and the continuing popularity of wildlife hunting as sources of protein are both desirable trends, as is reduced meat consumption in general. Diets low in red meat reduce the risk of death and disease, in addition to indirectly benefitting salmonid habitats.

Development of alternative energy sources could reduce dependence on hydropower and potentially allow for the removal of some hydroelectric dams. Energy conservation is a major source of new energy, but wind farms have considerable potential near the coast and in the Columbia Gorge. In addition, solar power and fuel-cell units in individual buildings are likely to become more popular as their unit costs decrease and hydropower rates increase.

Perhaps one of the most effective ways in which our culture could conserve salmonids and their environments is to remove many direct and indirect subsidies that encourage resource use and consumption. Many of these subsidies were initially intended to facilitate the development of the West, long before the environment was a significant societal concern, and they continue at substantial economic and environmental expense. For example, postal customers subsidize both the delivery of junk mail they do not want and its disposal in landfills. Taxpayers indirectly pay for building in high risk areas (floodplains, faults, fire-prone lands, ocean shores) through costs of fire suppression and disaster relief. Farmers are aided by taxpayers through drought and crop insurance and Federal price supports. In the West, the Bureau of Reclamation may have spent as much as $70 billion on water projects for agriculture since 1902 (DeBonis 1994). Automobile use is promoted rather than discouraged through subsidies to oil and gas industries that result in lower fuel prices—which encourages consumption—and through Federal and State fuel and licensing taxes that foster road improvements, which in turn stimulate more driving, taxes, and roads. Major electric power consumers are subsidized with lower rates for greater consumption, and the Bonneville Power Administration is subsidized by taxpayers, allowing it to provide extremely inexpensive electric power to its customers. Taxpayer subsidies on public lands are estimated at $700 million for below-cost timber sales and $95 million for below-cost grazing fees and wildlife control, and the patenting of public lands at $2.50 per acre of land for mining is tantamount to a substantial Federal subsidy (DeBonis 1994). In summary, all of these subsidies provide disincentives for conserving energy and resources, and in many cases promote excessive use and consumption. Obviously, many of these subsidies provide values that benefit some or all segments of society. But it should be made clear to the public that these programs have associated environmental costs and directly or indirectly influence the ability of aquatic systems in the Pacific Northwest to produce salmonids.

In addition to the above changes, we need to reconsider fundamental policies in four areas of our culture: population, economics, ethics, and education. The first three are the cultural forces that are the root causes of environmental degradation and salmonid extirpation. Education is the method by which we begin to change our minds, and of which this document and others like it are a part. Although these forces are closely interconnected, we list and discuss them separately.

8.10.2 Population Policy

Since 1870 the population of Oregon has doubled every 35 years, and the population of the Willamette Valley has tripled since 1940. The Pacific Northwest as a whole has experienced a population growth rate of 1.3% per year over the past decade, mostly in metropolitan areas (Matske 1993). Even at this slower rate, human population will double in 54 years and quadruple in 108 years. Growth will be concentrated in the current metropolitan areas, making the region even more urbanized than it is now. The salmon and spotted owl crises have made it abundantly clear that all the commodity, ecological, and aesthetic values desired by our culture cannot be met with our current and finite resource base. Thus, it is difficult to imagine how we can protect and provide these values with a population two- to four-times larger without substantial modifications in or use of water, power, food, fiber, open space, and fish. Randers and Meadows (1973) argue that the sooner we decide appropriate population levels, the more likely we will be able to choose the set of pressures we prefer to employ in stopping or reversing population and consumption growth, rather than have nature choose them.

It is also important to realize that the Pacific Northwest supports excessive populations elsewhere through exports of forest, agricultural, energy, and fishery resources. Carrying capacity and ecological integrity in our region are diminished by high population densities elsewhere in the United States and internationally. This is not to say that we should retain the resources so that we can maximize human population in this region, because the optimum human population is less than the maximum (Hardin 1968).

8.10.3 Economic Policy

As suggested above, there is a close relationship between human population size and per capita resource consumption. Ehrlich and Holdren (1973) describe this mathematically as $I = P \cdot F$, where I equals total impact, P is population size, and F is per capita impact. They add that this relationship is not necessarily linear, rather the factors are interconnected. For example, higher per capita resource consumption is associated with lower population growth rates and higher population size leads to greater per capita impact, especially among the very poor and very wealthy. The important issue is that, although many people consider human overpopulation a serious global issue, fewer accept that human overconsumption is equally serious. Both population and consumption must be considered to have a dramatic effect on impact because increases in either can offset decreases in the other. For this reason we spend a little more time discussing economic issues than population issues, and in relating the two.

As Packard (1960) described, the culture of the West seems centered around population growth and overconsumption of material things. However, his reflections are not new. In the Bible, Isaiah (44:14–20) discussed the difference of using wood to satisfy basic needs (warmth, cooking) and the surplus for making an idol (infinite wants), adding that his subject's inability to see the difference prevented him from correcting his error. The classical economist Mill (1857) wrote that "if the earth must lose the pleasantness that the unlimited increase of wealth and population would extirpate from it, merely to support a larger, but not a happier or better population, I sincerely hope they will be content to be stationary, long before necessity compels them to it." He added later that "a stationary condition of capital and population implies no stationary state of human improvement. There would be as much scope as ever for all kinds of mental culture, and moral and social progress . . . and much more likelihood of its being improved" (Mill 1857). Keynes (1931), like Isaiah, contrasted absolute needs or necessities that can be satisfied with insatiable needs that make us feel superior to others.

More recently, two imminent economists with global experience have argued for steady-state economies. Schumacher (1973) felt that the aim of an economy "should be to obtain the maximum of well-being with the minimum of consumption." He also argued that "production from local resources for local needs is the most rational way of economic life because dependence on distant imports and exports is highly uneconomic." Schumacher found modern economics inadequate for evaluating the value of air, water, soil, natural living organisms, natural ecological processes, beauty, health, cleanliness, and appropriate human lifestyles. He considered GNP misleading because it does not account for the destruction and depletion of natural and cultural resources. Schumacher wrote that "when the available spiritual space is not filled by some higher motivations, then it is filled by something lower." In particular he believes that "the acquisition of wealth and materials has become the highest goal, but we must develop a life-style that accords material things as secondary. The chance of mitigating the rate of resource depletion or of bringing harmony into relationships between those in possession of wealth and power and those without them is nonexistent as long as there is no idea anywhere of enough being good and more than enough being evil" (Schumacher 1973).

Daly (1973), following Mill and Schumacher, argued that both for the good of humans and the earth, economic growth should be in services and

leisure versus material goods. He described extra GNP in a rich country as satisfying relatively trivial wants because it mostly goes to the wealthier classes. The benefits of economic growth, he stated, go mainly to the rich, while the costs go mainly to the poor. He described current economic growth as being in increased output of goods per hour, meaning that the value of an hour rose in terms of goods. Daly concluded that as time becomes more expensive, fewer activities are worth the time, and time-intensive activities, like friendships, care for others, education, and meditation are sacrificed. Advertising stimulates this material consumption. Daly felt that a higher relative price of materials relative to leisure and services is needed. If raw materials are held constant and if all the ecological costs of production are included, so that costs of production increase instead of decrease, the incentive for producers to expand are eliminated (Daly 1973). He states that either producers or consumers of products must internalize the pollution and species-extirpation costs of production. Low production and consumption rates mean greater life expectancy of goods and people, less time lost to production, and less resource depletion and pollution (Daly 1973).

Increased consumption and economic growth results in decreased environmental quality because the production, use, and disposition of commodities creates environmental decay. This could be corrected somewhat if difficulties in accounting for externalities, optimizing the distribution of goods and income, and considering future generations were overcome (Barkley and Seckler 1972). Goods produced at high ecological cost could also be taxed highly (Durning 1994). Since the rate of economic growth is controlled by U.S. government officials, it can be slowed. In the long run, economic policies may have as much impact on salmonids as Federal forest plans or other more direct measures aimed at protecting Pacific Northwest ecosystems.

In various societies, economic surpluses have been used by priests that control religious sites to extract tribute, feudal lords that control land to obtain rents, and State and private capitalists that control capital to gain more capital. None of these institutions has nurtured our species to develop its greatest potentials and all have been highly destructive of our natural environment. If we hope to develop human potentials and to share the earth with anything resembling the current diversity of organisms and ecosystems present, we might be wise to listen to the economists who have long called for taking a different path than one that has historically led to ruin.

8.10.4 Ethics

The third major area in which fundamental change has been called for is in the way in which we view ourselves, others of our species, and our environment. Recently the UCS (1992) stated that "a new ethic is required—a new attitude towards discharging our responsibility for caring for ourselves and for the earth. . . . This ethic must motivate a great movement, convincing reluctant leaders and reluctant governments and reluctant peoples themselves to effect the needed changes." Although the right of property ownership holds great importance to citizens of the United States, all landowners have a responsibility to practice good stewardship to ensure that the activities in which they engage do not adversely affect resources that belong to all citizens.

Many may believe the call for a conservation ethic is new, but it began in the United States as early as the 1850's in the writings of Thoreau (Nash 1989). Thoreau writes that "What we call wildness is a civilization other than our own," and "There is no place for man-worship . . . take wider views of the universe." Both in his simple life and in his writings Thoreau treated nature as an equal and implied that it should have legal rights similar to those of other minorities oppressed by the dominant culture. Later, Marsh (1864) writes that human stewardship of nature is a moral issue, not just an economic one, and he was very concerned with consumption and waste of natural resources. John Muir's journals also included references to the "rights of animals" and "the rights of all the rest of creation" (Nash 1989). The major scientific basis for natural rights for nature lies in Darwin's (1859, 1871) works. Darwin demonstrated that the process of natural selection linked all past and present species, and challenged the idea that nature exists solely to serve humans. Darwin believed in kinship and respect for our fellow organisms, and argued that moral sympathies and ethics had survival value and thus were a product of natural selection. He wrote that ethics are the basis of animal societies and of the evolution from human families and tribes to nations and international organizations.

Most of the world's great religions and cultures included liberal views on natural rights. For example, Buddha emphasized a reverent and nonviolent attitude toward all sentient beings and especially trees, expecting observers to plant and nurture them (Schumacher 1973). The early Greeks and Romans felt that humans should respect nature, and suffer when they did not (Nash 1989). Writings of Hinduism, Judaism, and Taoism include similar guidelines and warnings on the proper way in which humans should interact with nature. Thus, more modern stirrings of conscience have a long history.

Modern thought on environmental ethics was stimulated by the development of the discipline of ecology and the writings of Schweitzer and Leopold.

Ecology quantified the interdependence between plants, animals (including humans), and their environments. Schweitzer felt that proper conduct for humans was based on a reverence for life. He wrote that an ethical person "shatters no ice crystal, tears no leaf, and crushes no insect" and that killing was done only when "absolutely necessary to enhance another life and then only with compassion" (Nash 1989). He received the 1952 Nobel Peace Prize. Leopold combined ethics and ecology, writing that "conservation based solely on economics is hopelessly lopsided because it ignores elements lacking commercial value but that are essential to healthy ecological functioning. It tends to relegate to government many functions too large, complex, and dispersed to be performed by government. An ethical obligation on the part of the private owner is the only remedy" (Leopold 1949). In perhaps his most eloquent phrase, he stated "a thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community. It is wrong when it tends otherwise."

Despite a long history involving many perceptive thinkers, the extension of natural rights to nature is an incredible and revolutionary thought to many people, but it is a very compelling one to many others (Nash 1989). The same incredulity, however, met the first proposals for granting rights to human groups throughout Western history. Each of the advances listed in Table **8-3** required substantial changes in the perspectives of the dominant social group. It can be argued, however, that the greater society is improved by such changes.

8.10.5 Education

Current economic thought stresses the importance of human creativity in developing a healthy economy. Older views of natural resources and labor as the only sources of capital are being expanded to include the human intellect. With this in mind, Schumacher (1973) considered education as the most vital of all resources. However, because a purely technological education limits the ways in which we can think and our views of ourselves as a species, he felt we need a rich mix of science and humanities education throughout our lives. Apparently this position is shared by a number of educators because current pedagogical approaches integrate science and humanities instruction in growing numbers of our public schools and universities. Also, many persons at both extremes of the political spectrum decry the absence of instruction on values, ethics, and the proper role of a conscientious citizen in human society and the larger ecosystem.

Environmental education, as taught in public schools, has typically consisted of descriptions of environmental problems. This is a necessary but insufficient step. What is needed is a citizenry that is aware of 1) the status and trends in our environment, 2) the physical and social causes of both, and **3)** the individual and social changes needed to place us on a sustainable path. Most importantly, we must become willing and motivated to make those changes. Just as public, university, and adult education played critical roles in expanding civil rights to all U.S. citizens, it can play a similar role in affecting how we view our environment and our role in it. Unfortunately, we have few models of sustainable resource use from which to draw, and these are largely restricted to situations where human population density was quite low by today's standards.

We in the Pacific Northwest, blessed by some of the most diverse and least disturbed ecosystems in the conterminous United States, have a rare opportunity. We can lead the rest of the Nation along a more sustainable path, or we can follow the

Table 8-3. Development of civil and natural rights in American and Western culture. Modified from Nash 1989.

Civil or natural group	Enabling legislation or doctrine
Israelite tribesmen	Ten Commandments (2500 BP)
English lords	Magna Carta (1216)
European American men	Declaration of Independence (1776)
Livestock	Martins Act (1822)
African Americans	Emancipation Proclamation (1863)
European American women	Nineteenth Amendment (1920)
Native Americans	Indian Citizenship Act (1924)
African Americans	Civil Rights Act (1964)
Endangered plant and animal species	Endangered Species Act (1973)
Natural Ecosystems	???

heavily trodden path that has emanated from western Europe and crossed this continent in a little over 500 years. We can lead the United States to a new consciousness (just as we have with air and water quality criteria, land-use planning, and recycling), or we can hope that the same institutional and ethical approaches that led us into our current resource crises can extricate us. We can choose a lifestyle that includes salmon and the ecosystems that support them, or we can continue to extirpate them both. Because we still have wild salmon, we still have a chance to save them. Because those salmon occupy an important position in our regional culture and thinking, range throughout the region, and are affected by nearly everything we do with the lands and waters, more than any other symbol they may force us to rethink the wisdom of our current culture.

This is not a new dilemma; it is as old as civilization. "How did civilized man despoil his environment? He cut or burned the timber from the hillsides and valleys, he overgrazed and denuded the grasslands with his livestock, he killed most of the wildlife, fish, and other aquatic life, he allowed his farm topsoil to erode and clog his waterways, he wasted the easily mined minerals, then his civilization declined or he moved it elsewhere" (Dale and Carter 1955). Since we cannot move elsewhere, we can either change our minds or watch the steady degradation of our resources and quality of life. The decisions we make over the next few years will govern the world we and our progeny inherit. Education is one key way in which we can begin to include present and future citizens in the decision-making process.

8.11 Summary and Implications for Salmonids

Because the health of native salmonid populations and the condition of their habitats are inextricably linked to so many aspects of resource extraction and use, protection and long-term recovery of these fish will require fundamental changes in management practices at many levels—improving fishery management and fish stocking practices, modifying land- and water-use practices, and ultimately addressing the root cultural aspects that drive the demand for natural resources. Fundamentally, effective management systems for improving salmonids and their habitats are those that, to the greatest degree possible, minimize disruption of natural ecological processes and mimic the extent and frequency of natural disturbance. Wild native fish assemblages should be encouraged to replace non-native stocks and species. Modification of fish harvest practices (e.g., reduced harvest levels, terminal fisheries, gear and angling restrictions) can help ensure adequate recruitment of spawners and minimize other ecological effects of harvesting (e.g., size selectivity, loss of nutrient inputs to streams from carcasses). Maintenance and re-establishment of natural channels and floodplain processes can be attained through active removal of humans structures, such as dams and levees, and by returning streams to more natural flow regimes.

Channels, riparian areas, and floodplains are unlikely to recover without modifying land-use systems to some degree, both in riparian and upland areas. Regardless of the land-use type, most impacts on salmonid habitats relate to the removal of vegetation and disturbance to soils, which lead to changes in the rate of delivery of water, sediment, organic debris, and nutrients to streams. Practices that limit the areal extent, frequency, and intensity of disturbance are likely to have the least impact on salmonid habitats. Impacts of forest practices can be substantially reduce by careful layout of harvest units, roads, and skid trails. Those practices that mimic the extent and frequency of deforestation from natural disturbances, such as fire, are most likely to support the hydrological and geomorphological processes that sustain healthy channels and riparian systems. On rangelands, livestock numbers, composition, and distribution, as well as the timing and duration of grazing, can all be controlled to ensure adequate vegetation remains on site to minimize erosion and hydrologic changes. Marked improvements in riparian vegetation and instream habitat conditions can be achieved by excluding cattle from the riparian zone, or by carefully controlling the timing and intensity of grazing. Increased amounts of permanent vegetative cover are also needed on agricultural lands to keep soil and chemicals out of streams. Mined areas can be restored to natural contours and vegetative cover, and contamination of streams with mining effluents can be reduced with containment and treatment. Urban land can be restricted from expanding into relatively undisturbed systems through zoning and higher-density housing.

Salmonids are also likely to benefit from increased planning at the regional level. To an increasing degree, State and Federal resource management agencies are developing cooperative programs for salmonid conservation and restoration; this coordination of effort is essential for addressing conservation at the watershed, basin, and region levels. And finally, conservation of salmonids fundamentally comes down to the behavior and actions of individual citizens. Simpler, less consumptive lifestyles and lower reproduction rates lead to reduced demand for resources, the extraction of which directly or indirectly affects salmonid

habitats. Residents of the Pacific Northwest are revealing increased concern with the survival of salmon and the goals of our civilization. Communities in the region can control their futures by establishing management systems based on a vision of desirable communities, landscape condition, and fish and wildlife populations—or they can allow continued unrestrained development at the expense of aquatic ecosystems and the salmonids they support. Already the landscape of the region has changed markedly. This trajectory can be reversed or accelerated only by the aggregate desires of its citizens. Humans are the critical variable in ecosystem management.

9 Relevant Federal Laws for Protecting and Restoring Salmonid Ecosystems on Nonfederal Lands

Three bodies of Federal law are most often cited in reference to the protection and restoration of salmonids and their habitats. The Clean Water Act (CWA) commonly refers to a group of laws intended to protect the quality and biological integrity of the Nation's waters (codified as 33 USC 1251 *et seq.* and 33 USC 1311 ***et*** *seq.* are the Act itself [CWA, PL 92-500 19721 also known as the Federal Water Pollution Control Act [FWPCA 19481 as amended by CWA [PL 92-500 1972 and PL 95-217 19771, and the Water Quality Act [WQA, PL 100-4 19871). The National Environmental Policy Act (NEPA, codified as 42 USC 4321 *et seq.*, passed as PL 91-190 1970, and its amendment the Pollution Prosecution Act [PPA, codified as 42 USC 4321, passed as PL 101-593 19901) as well as the Endangered Species Act (codified as 16 USC 1531 *et seq.*, passed as PL 93-205 1973) addresses habitats and organisms.

A fourth, less well known law, the Food Security Act (FSA, bound as 99 Stat 1354, passed as PL 99-198 1985) seeks to discourage land uses in sensitive areas (i.e., erodible soils and wetlands) through controls on Federal assistance programs. The goals and certain sections of these Federal laws that explicitly pertain to Federal and nonfederal landowners are discussed in the sections that follow. Consequently, this chapter provides only a brief overview of these laws; an exhaustive review of their effectiveness in protecting and restoring salmonids exceeds the scope of this document.

9.1 Clean Water Act (CWA)

The goals and policy of the CWA are to restore and maintain the chemical, physical, and biological integrity of the Nation's waters; to eliminate the discharge of pollutants into waters; to attain water quality that provides for the protection and propagation of fish, shellfish, and wildlife; and to develop and implement area-wide waste treatment management to control pollutant sources.

Several sections of the CWA pertain to habitat restoration. Section 2 requires States to identify areas with substantial water quality control problems (§§ 2, 31, 32, 33(a), 34 of the CWA; § 35 of FWPCA; §§ 101(d) and 101(e) of WQA [codified as 33 USC 1288])[2]. After consulting with local governments and contiguous States, States shall develop effective area-wide waste management plans for all wastes in the area. The plans shall identify and set forth controls for municipal, industrial, agricultural, silvicultural, and mining wastes to protect ground- and surface-water quality. The first sentence of Section 303 (WQA, codified as **33** USC 1311) directs States to promulgate water quality standards that protect fish and wildlife. It also obligates States to estimate total maximum daily loads for pollutants to assure protection of balanced indigenous populations of fish, shellfish, and wildlife. The WQA defines pollution as the man-induced alteration of the ecological integrity of water. This section (33 USC 1311) also addresses the illegality of pollutant discharges: "the discharge of any pollutant by any person shall be unlawful," and subject to penalties. Section 2 (CWA; § 308(d) of WQA; 33 USC 1313) directs EPA to issue guidelines for controlling pollution from agriculture, silviculture, mining, construction and hydrological modifications, and for restoring and maintaining ecological integrity in receiving waters. Section 2 (CWA; § 305 of WQA; § 52 of FWPCA; 33 USC 1315) requires States to prepare biennial reports describing the quality of all State waters; the degree to which those waters provide for the protection and propagation of balanced populations of shellfish, fish, and wildlife; and additional actions needed to achieve such objectives. Finally, Section 401 of the original 1948 law (§ 2 of CWA; §§ 61(b) and 64 of FWPCA; 33 USC 1341) directs Federal permittees involved with activities resulting in a discharge to certify that the discharge will comply with water quality standards.

'Agencies often refer to this section as "Section 208," a number derived from a version appearing as a committee print. See: U.S. Senate, 1977, Committee on Environment and Public Works, *The Clean Wafer Act and Changes Made by the 1977 Amendments,* 95th Congress, 1st Session, Committee Print 95-12, p. 34-41, Washington, **DC:U.S.**Government Printing Office. The Agencies also refer to Sections 303, 304, 305, etc. from this committee print which became parts constituting Section 2 of CWA (33 USC 1251 *et seq.).*

The **CWA** has been relatively effective in reducing pollution from point-source industrial and municipal discharges but has been much less effective in controlling nonpoint-source pollution, preventing cumulative effects on water quality, or protecting streams from habitat degradation unrelated to contamination (e.g., modification of hydrologic regimes, alteration of stream channels, introduction of non-native species) (Karr 1990; Doppelt et al. 1993; Karr 1995). In this regard, the goal of the CWA of maintaining biological integrity of the Nation's surface waters has not been met (Hughes and Noss 1992). The 1987 amendments to the CWA contain stronger provisions for controlling nonpoint-source pollution and place greater emphasis on protecting instream biological resources (Doppelt et al. 1993). In addition, EPA has begun a program to better protect aquatic biota. States have been directed to develop narrative biological criteria for streams, rivers, lakes, wetlands, and marine ecosystems (EPA 1990). Karr (1995) concluded that establishment of biological criteria (rather than solely chemical criteria) is essential for ensuring long-term protection of aquatic ecosystems.

9.2 National Environmental Policy *Act* (NEPA)

The National Environmental Policy Act (42 USC 4321 ***et seq.***) comprises two laws, the original NEPA (PL 91-190, passed in 1969 but dated 1970) and the Pollution Prosecution Act (PL 101-593 1990). Section 2 of NEPA (42 USC 4321) declares its purposes: to declare a national policy that encourages harmony between humans and their environment, reduces environmental damage, and improves understanding of ecological systems. NEPA recognizes the impact of human activity on the natural environment, particularly the profound influences of population growth, urbanization, industrialization, resource exploitation, and technology (§ 101; 42 USC 4331). The Federal government is responsible for coordinating Federal programs to assist each generation of the Nation to act as trustees for future generations, preserve a diverse environment and important natural aspects of our national heritage, and maximize the recycling of depletable resources. This policy section also recognizes the rights and responsibilities of each person to enjoy, preserve, and enhance the environment.

Section 102 of NEPA (42 USC 4332) requires all Federal agencies to administer laws and regulations in accord with the above policies and to give appropriate consideration to unquantified environmental amenities. For major Federal actions and legislation, all Federal agencies are obligated to provide a detailed statement on the environmental impact of the proposed action or legislation, any unavoidable adverse environmental effects, and alternatives. The agencies are also required to use ecological information in resource planning and development projects and to make information available to nonfederal institutions for restoring and maintaining environmental quality. In 1978, the Council on Environmental Quality published regulations to implement NEPA (40 CFR 4321 *et seq.* 1969, 1970). These regulations, among other things, require environmental analyses to consider cumulative effects, which are defined as "the impact on the environment which results from the incremental impact of an action when added with other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or nonfederal) or persons undertakes such other actions." (Beschta et al. 1995). Judicial interpretation of the cumulative- effects language of NEPA has determined that agencies must consider impacts resulting from both Federal and private actions in determining cumulative effects within the area of a proposed action (Doppelt et al. 1993).

Because NEPA requires the analysis of cumulative effects, the Act has pushed the agencies toward watershed-level (or broader) assessment of environmental impacts (Doppelt et al. 1993), which should foster greater protection of aquatic ecosystems. However, the Act does not guarantee that environmental impacts of an action will be avoided or mitigated. It only requires that alternatives to the action be considered, that a thorough analysis of the expected environmental impacts associated with each alternative be performed, and that these impacts be disclosed to the public. Final decisions do not require that the identified impacts be avoided, thus environmental degradation can still occur. In addition, alternatives sometimes differ little from one another or are selected for political rather than ecological reasons. Finally, NEPA is usually directed toward individual projects rather than toward far-reaching policies. In particular, the Act has not been used to develop coordinated economic, ethical, or population policies that ultimately govern environmental quality.

9.3 Endangered Species Act (ESA)

Congress found that various species have become extinct in this country as a result of economic growth or development and that those same forces threaten or endanger other species. Section 2 **(16** USC 1531) of ESA states its purposes: "to provide **a** means whereby the ecosystems upon which threatened and endangered species depend may be conserved and to provide a program for the conservation of such endangered species and threatened species.. . ." The final purpose is to honor the Nation's environmental

treaties and conventions. Moreover, ESA explicitly describes two policies: all Federal agencies are to seek conservation of listed species and to cooperate with nonfederal agencies in resolving water resource issues in concert with the conservation of endangered species. Six sections are especially pertinent to the protection and restoration of salmonids and their ecosystems.

Section 3 (16 USC 1532) defines several terms in use throughout the Act. Critical habitat includes all areas occupied by the species as well as unoccupied areas essential for species conservation. A species includes subspecies and distinct population segments that interbreed when mature (e.g., salmon stocks). Take includes harassment, harm, pursuit, trapping, collecting, capture, or any attempt to do so. Harassment has been further defined to mean the intentional or negligent act or omission that significantly disrupts normal behavior patterns of the endangered or threatened species. Harm can include activities that result in significant environmental modification or degradation of the habitat of an endangered or threatened species.

Section 4 (16 USC 16 1533) describes the listing process. A petitioned species may be listed as endangered or threatened because of present or threatened modifications to its habitat or range, overexploitation, disease, predation, or other factors. Within 90 days of receiving a petition, the agency must publish whether such an action may be warranted. If so, a status review begins, and within 12 months of the petition the agency must publish its decision. Those species most likely to conflict with economic activities are prioritized for listing, which is based solely on the best scientific data available. Negative findings are subject to judicial review. If listed, the agency issues recovery plans and regulations to conserve the species and prohibit take or violations of the regulations. Recovery plans include critical habitat designations (that include consideration of the economic impact of the listing), necessary management actions, and objective measurable criteria for assessing status and trends for at least five years.

The remaining four sections involve implementation of recovery plans. Maximum cooperation with States is encouraged by Section 6 (16 USC 1535). Section 7 (16 USC 1536) obligates all Federal agencies to minimize placing listed species in further jeopardy and to perform biological assessments. Section 9 (16 USC 1538) makes it unlawful to take, possess, or violate any regulation pertaining to a listed species without a permit. Section 10 (16 USC 1539) outlines permit conditions. Otherwise prohibited acts are permitted for scientific purposes, to enhance species persistence, or if take is incidental to lawful activity. Issuance of such a permit requires a conservation plan that specifies likely impacts, steps and funding to mitigate such impacts, reasons alternative actions are not taken, evidence that the taking will not reduce species persistence and recovery, and responses to other requirements of the agency. The agency may also allow permits for a year or less if human subsistence or substantial economic losses are imminent. All permits are subject to public hearings.

The ESA has been one of the Nation's most powerful conservation acts. However, although the language of ESA emphasizes protection of habitats and ecosystems, the Act has not been interpreted to ensure proactive land management. Species listings generally occur after populations have substantially declined and their habitats have been greatly altered or degraded (Doppelt et al. 1993). Consequently, by the time a species is listed, options for recovery are often limited, and the costs associated with restoration may be high. In addition, ESA does not require review of projects that endanger entire ecosystems (Karr 1990). And finally, few species that have been listed under the Act have recovered sufficiently to be delisted; for example, "not a single fish warranted removal from the (American Fisheries Society) list because of successful recovery efforts" (Williams et al. 1989). Thus, while ESA has been useful in curbing losses of listed species and their habitat, the Act by itself is not likely to prevent general habitat degradation and additional species listings. Doppelt et al. (1993) provide a more thorough discussion of the benefits and shortcomings of ESA in protecting aquatic ecosystems.

9.4 Food Security Act (FSA)

A fourth law is especially relevant to private landowners. Within the Food Security Act (bound as 99 Stat 135, passed as PL 99-198 1985), Title XII—Conservation contains two subtitles that directly affect nonfederal landowners. Subtitle A—Highly Erodible Land Conservation (§§ 1211–1213) stipulates that any person who in any crop year produces a crop on highly erodible land shall be ineligible for any type of price support, payment, loan, crop insurance, or disaster payment. Exemptions to this section are granted only to persons actively applying conservation plans approved by the local soil conservation district. Subtitle B—Wetland Conservation (§§ 1221–1223) states that any person who in any crop year produces crops on converted wetlands shall be ineligible for any type of price support, payment, loan, crop insurance, disaster or payment. Landowners contract with the U.S. Department of Agriculture to implement a conservation plan, convert the land to its planned use, and agree not to conduct any

harvesting, grazing, or tree planting unless allowed in the contract. This law provides incentives for farmers, ranchers, and silviculturalists to conserve riparian areas and wetlands—essential components of salmonid habits—to continue receiving Federal subsidies.

9.5 Summary and Conclusions

Three circumstances justify examining Federal laws in this technical foundation document. First, certain laws require decisions to be made based on scientific information or scientific research. Second, other laws mandate technical standards or scientific guidelines that must be met. Third, the laws outline management strategies that require scientific information or data to make decisions. Scientific knowledge can be incorporated into regulations pursuant to law or the law itself; however, if the scientific basis is not understood or is controversial, committees or commissions can serve formally to assess applicable state-of-the-science.

In addition to the body of law presented here, other Federal laws explicitly extend Federal help to nonfederal, private landowners. One example is the Forest Stewardship Act (PL 102-574 1992, codified as 16 USC 2101), an amendment to the Federal Forest and Rangeland Renewable Resources Planning Act (PL 94-588 1976, codified as 16 USC 1600 *et seq.*), which authorizes the Secretary of Agriculture to assist States and local foresters in establishing a coordinated, cooperative stewardship program for management of nonfederal forest lands and "the improvement and maintenance of fish and wildlife habitat" (16 USC 2101[b][6]).

Clean water law, NEPA, **ESA,** and **FSA** provide substantial Federal leadership and funds, including scientific information, to the States and local landowners to accomplish explicitly stated goals. These laws depend on scientific information to accomplish their objectives. Each act contains language linking human and ecological values, recognizing that ecologically healthy, biologically diverse environments provide healthy physical and economic environments for people.

10 Monitoring Aquatic Ecosystems

Despite the considerable effort and expense devoted to management of natural resources, including salmonids and their habitats, management activities rarely are accompanied by rigorous monitoring programs to determine whether plans are being implemented as designed and whether they are having the desired effects. The periodic reappraisal of management activities—where information is gathered to assess progress towards management goals and to redefine those goals if necessary—forms the backbone of the "adaptive management" philosophy. Many researchers and institutions have called for increased monitoring of the effects of land management activities at spatial scales ranging from sites and watersheds to basins and regions (Karr in press; FEMAT 1993; NRC 1992; GAO 1981, 1986).

Although there are many kinds of monitoring, two types are central to a conservation strategy for salmonids: implementation (or compliance) monitoring and assessment (or effectiveness) monitoring. Implementation monitoring involves determining if standards, guidelines, or prescriptions of a particular plan or program are being followed. Assessment monitoring is intended to evaluate whether implementation of the plan or program is achieving management objectives. A review of these two types of monitoring is instructive of the pitfalls likely to befall implementation of a salmonid conservation strategy.

10.1 Examples of Existing Implementation (Compliance) Monitoring Programs

With respect to salmonid conservation activities, the purpose of implementation monitoring is to determine if a landowner is correctly applying the standards, guidelines, or prescriptions designed to protect and restore aquatic and riparian ecosystems. Specific standards are likely to vary with the nature of the activity and the particular species of concern. Such standards might ensure maintenance of adequate riparian buffers, avoid sensitive hillslopes, conform roads to a watershed road plan, prescribe structures (e.g. fences, settling basins for sediments) be built to specifications, and verify elements of a watershed analysis. Monitoring for compliance with conditions identified in these plans could involve remote sensing of the management activity as well as site visits. Part II of this document provides a detailed discussion of elements likely to be included in a habitat conservation plan, as well as a suggested approach for compliance monitoring.

There are at least four Federally mandated, State-implemented, compliance-monitoring programs that provide insights into the development and implementation of sound monitoring for salmonid conservation practices. These are the Environmental Protection Agency's (EPA) National Pollution Discharge Elimination System (NPDES), the U.S. Army Corps of Engineer's Section 404 wetland mitigation and permitting program (WMPP), the Forest Services's (FS) Best Management Practices program (BMP), and the EPA's Rural Clean Waters Program (RCWP). Each of these programs has recently been reviewed and limitations identified.

NPDES is a permitting process for regulating emissions of pollutants from point-source facilities. The NPDES process requires monitoring pollutants characteristic of the particular type of discharge. This means that few facilities monitor the same constituents. An examination of chemistry data available from industries discharging waste into the Willamette River, Oregon, found that no quality assurance information (blanks, replicates, calibration standards, reference standards) was provided, which prevented quantitative assessment of data quality (TT 1992). Detection limits occasionally were not reported. Other problems included inconsistent permit requirements, variables, analytical techniques, and reporting units, as well as a lack of an electronic database. Such data, whether from point or diffuse sources, have poor utility for quantitative evaluations of the degree to which dischargers treat effluent.

WMPP is an outgrowth of the Clean Water Act requiring mitigation for "unavoidable" wetland destruction by construction of a new wetland of similar size and type to ensure no net loss of wetlands. The WMPP's of several States were evaluated by a group of collaborators, namely, Gwin and Kentula (1990), Gwin et al. (1991), Holland and Kentula (1992), Kentula et al. (1992), and Sifneos et al. (1992a, 1992b). These researchers concluded that 1) objectives of created wetlands are frequently unclear; 2) project plans often indicate unfamiliarity with the literature and past failures as well as poor

understanding of the surrounding landscape; 3) projects are rarely designed or constructed as required in the permits; 4) data collection and storage are generally inadequate for effectively assessing compliance (particularly for area affected, vegetation, and hydrology), and 5) implementation monitoring is insufficient. They called for staffing by professional ecologists, specifying objectives and numerical criteria, increased monitoring and verification at all project phases, and computerized databases. They added that monitoring should be sufficient to determine compliance with specific permit requirements, as well as to evaluate the effectiveness of the project in restoring wetland elements and processes (assessment monitoring). Area-wide reports on patterns and trends in implementation should be regularly produced.

The Forest Service's best management practices (BMPs) are intended to restore and protect streams by facilitating natural riparian vegetation condition and slowing sediment delivery. BMPs were evaluated for the Clearwater National Forest in Idaho by Rhodes et al. (1994), who found that the guidelines were too general to be effectively implemented and evaluated, and they often were contradictory. Moreover, guidelines were not framed within an ecosystem or watershed approach. The 1976 forest plans also lacked an ecological context as did the models from which they were generated. Unrealistic timber targets and recovery assumptions resulted. Similarly, the modeled trends for road recovery, sediment delivery, and substrate sedimentation did not agree with observed conditions either before or after the treatments. Rhodes et al. found, therefore, that the models only gave the illusion of rigor. Riparian timber harvest schedules were violated, partly because the extent of riparian areas was underestimated. Water quality standards were considered too obtuse to be usefully assessed and too permissive to protect fishery resources. Post-harvest timber recovery was assumed, but inadequately monitored because of the subjective, qualitative, and cryptic nature of the data. In addition, criteria were lacking altogether for monitoring and assessing watershed condition, fish habitat, and fish populations (assessment monitoring issues). The FEMAT (1993) and PACFISH (FS and BLM 1994a) standards and guidelines were written to correct many of these shortcomings.

The Rural Clean Water Program was a Federally cost-shared nonpoint source pollution control effort to improve water quality in 21 watersheds across the country. Projects included Tillamook Bay, Oregon, and Rock Creek, Idaho, and all involved U.S. Department of Agriculture and EPA participation together with State and local coordination and monitoring. In their evaluation of the program, Gale et al. (1993) list several lessons relevant to compliance monitoring. A thorough program evaluation by technical experts should be planned, funded, and scheduled at its initiation. Technical assistance, workshops, and periodic onsite evaluations should be included. Standardized, streamlined, annual, and final reports of activities and areas affected should be required. At the beginning, specific and measurable objectives must be set at the watershed level by representatives from the community and project agencies. These objectives should reflect desired outcomes but allow modification with increased knowledge. A computerized database, aerial photographs, and geographic information system are effective tools for tracking and reporting on project implementation, but regular visits to landowners reduce the number of misunderstandings.

The above examples suggest that a successful implementation monitoring program should consist of several key elements. Foremost, implementation monitoring programs must be adequately funded and staffed by ecologists with experience in geomorphology, hydrology, soils, vegetation ecology, fisheries ecology, database management and GIS, and geography. Specific objectives, project specifications, and tracking criteria must be included in plans, and remote sensing and periodic site visits are essential. Information will be most useful if it is entered into a computer database, using standardized streamlined forms, portable data recorders, or both. Periodic status and trend reports should be produced and the program should be technically re-evaluated every few years. The NRC (1992) concluded that many restoration projects failed because project specifications were ignored, insufficient ecological knowledge was incorporated in the planning and installation, specific objectives and criteria for tracking and redirection were lacking, and pre- and post-evaluations were omitted.

10.2 Examples of Existing Assessment (Effectiveness) Monitoring Programs

The purpose of assessment monitoring relative to salmonid conservation planning is to determine the degree to which permit compliance results in maintaining or improving habitat conditions and salmonid populations. It is essential that this connection be made because the goals of habitat conservation plans are to protect and restore salmonids. As with implementation monitoring, much can be learned from the strengths and weaknesses of other assessment monitoring programs.

Both State and Federal agencies currently conduct assessment monitoring programs for forestry,

fishery, and other water resources. Many management activities, however, are not monitored at all. For example, Frissell and Nawa (1992) indicated that large numbers of artificial habitat structures have been placed in Pacific Northwest streams without any serious monitoring program to evaluate their efficacy. There are no systematic State or multistate monitoring programs in place at present; consequently, monitoring results only have site-specific or basin-level applicability at best.

The States of Idaho, Oregon, and Washington all have programs and written protocols for monitoring salmonids, salmonid habitats, water quality, and benthos (Table 10-1). Only one California protocol was located, so knowledge of that State's methods is incomplete. There is considerable comparability in parameters and methods among the States of EPA Region X (Idaho, Oregon, Washington). This is largely a result of three parallel developments. Biologists from the water quality agencies began working in 1990 to develop and test a common set of bioassessment and physical habitat protocols, culminating in publication of the biological monitoring handbook of EPA's Region X (Hayslip 1993). In addition, California, Oregon, and Washington are conducting regional EMAP (EPA's Environmental Monitoring and Assessment Program) surveys that use common field protocols. State fishery agencies in Oregon and Washington have also developed comparable modifications of Hankin and Reeves' (1988) methodology through cooperative work with the Forest Service (FS), Bureau of Land Management (BLM), and Washington's Timber, Fish and Wildlife program. In addition, Oregon has conducted since 1990 a stratified, random-sampling survey of the spawning habitat of coho salmon in coastal streams (Jacobs and Cooney 1993).

Several Federal agencies have begun assessment monitoring programs for streams in the region. The Geological Survey characterizes basins, segments, and reaches through a subjective site selection process and on a nine-year rotation. The FS and BLM inventory wadeable streams with a projected return interval of ten years, but the watershed selection is subjective. The National Biological Service (NBS) aggregates available biological data and published information to produce periodic reports. The EPA has initiated regional stream monitoring based on a statistical sampling survey design, a four-year return interval, annual sampling, and annual revisits to a subset of sites.

At the monitoring-design level, several of these programs present serious shortcomings to assessment monitoring of Federal and nonfederal lands and waters as it relates to salmonid conservation. Subjective site selection precludes use of inferential statistics and extrapolation to unsampled sites, making a regional or multistate assessment impossible. Instead, the data are only applicable to the sites selected, and they are prone to selection biases and statistical inconsistency. In addition, return intervals at the decadal scale hinder trend detection, which is a function of the rate of environmental change relative to the number of years the population is sampled. Stevens (1994), Larsen et al. **(1994,** 1995), and references cited therein provide further explanation of these issues. Aggregations of available data, such as those of the NBS, suffer from the limitations of the sampling designs from which they were produced (mostly subjective) as well as differing indicators and sampling methods. At best these aggregations produce qualitative information on site-specific conditions; they cannot be used for quantitative estimates of status or trends in aquatic ecosystems or biota. The inventory approach favored by the FS and BLM requires walking all stream reaches, and thus provides considerable site-specific information about each one. This level of effort is extremely expensive and time consuming; it is also more likely to employ qualitative indicators that are much less sensitive to trend detection than quantitative indicators.

There is somewhat more consistency among the Federal monitoring programs at the reach scale. All three field programs use ten or more transects selected in a randomized systematic manner from which samples are taken. In addition, common indicators are used, but these differ in rigor (Table 10-2). Each program has its strengths ,and weaknesses, but greater quantification and increased completeness of indicators raises the likelihood of accurate and precise assessments of status and trends.

There is a clear need for all Federal and nonfederal institutions that are monitoring salmonid ecosystems to adopt a common sampling design, indicators, and sampling protocol. Indicators should be based on quantitative measurements to the greatest degree possible to reduce measurement variance and to provide early detection of trends. Probability-based (e.g., randomized systematic) sampling designs should be implemented to facilitate regional or basin-level population estimates, and to ensure data are collected in the most cost-efficient manner.

10.3 Sampling Design Considerations

As noted above, temporal and spatial trends in the condition of aquatic ecosystems can be most effectively assessed using a systematic, randomized sampling design. Several recent studies document the efficiency of random sampling compared with subjective sampling of aquatic systems. Landers et al. (personal communication) demonstrated that results from subjectively chosen sites differ from those obtained from randomly selected sites. They

Table 10-1. Monitoring parameters of Pacific Northwest States*

Parameters	California	Idaho	Oregon	Washington
Temperature (recording)	✓	✓	J	J
Dissolved oxygen	✓	✓	✓	J
Conductivity	J	J	✓	✓
pH	J	J	✓	✓
Statistical sampling design (sitelstation)	✓/✓	/✓	✓/✓	✓/✓
Qualitative				
Percent fines		J	✓	✓
Embeddedness	✓	✓	J	✓
Fish cover	✓	J	J	J
Velocity/depth		✓	J	J
Channel shape		J	✓	J
Pool/riffle		✓	J	J
Width/depth		✓	J	J
Bank stability		✓	✓	✓
Bank vegetation		✓	J	✓
Riparian buffer		✓	J	✓
Stream disturbance		J	J	J
Pool character		J	J	J
Winter refugia		J	J	J
Canopy cover		J	✓	✓
Stream/valley type		J	J	J
Channel sinuosity		J	J	J
Habitat units		✓	✓	✓
Quantitative				
Discharge	✓	J	J	✓
Depth and widths	✓	✓	J	✓
Gradient	✓		J	✓
Bottom substrate	J	✓	J	✓
Large wood	✓		✓	✓
Residual pool depth	J	J	J	✓
Insolation		J	J	
Canopy closure (densiometer)	✓	✓	✓	✓
Bank character	J	J	✓	✓
Benthos (quantitative)		J	J	✓
Salmonid spawning	J	J	J	✓
Fish (quantitative)	J	J	J	✓
Reference sites (regional)		J	J	✓

* From Ralph (1990), Burton et al. (1991), Cowley (1992), Chandler et al. (1993), Clark and Maret (1993), Hayslip (1993), MSG (1993), Runyon (1994), Schuett-Hames and Pess (1994), and Schuett-Hames and Pleus (1994).

Table 10-2. Reach-level monitoring parameters of Federal Programs in the Pacific Northwest*

Parameters	Forest Service1 Bureau of Land Management	U.S. Environmental Protection Agency (EMAP)	U.S. Geological Survey (NAWQA)
Statistical sampling design (sitelstation)	/✓	✓/✓	
Regional reference condition		J	J
Algae (optionallcore)		✓/	/✓
Benthos (optionallcore)	✓/	/✓	/✓
Fish assemblage (optionallcore)	✓/	/✓†	/✓
Salmonid spawning (optionallcore)	✓/	✓/	
Riparian bird assemblage (optionallcore)		JI	
Microbial respiration (optional/core)		✓/	
Major cations and anions (quantitative)		✓	
Nutrients (quantitative)		✓	J
Iron and manganese (quantitative)		✓	J
Turbidity and color (quantitative)		J	✓
pH and conductivity (quantitative)		✓	✓
Dissolved oxygen (quantitative)		J	J
Temperature (quantitative)	✓	J	J
Depth and width (quantitative/qualitative)	✓	✓/	✓/
Habitat type (quantitativelqualitative)	✓	/✓	/✓
Large woody debris (quantitativelqualitative)	✓	✓/	
Fine sediment (quantitative/qualitative)		✓/	
Bank height (quantitativelqualitative)		JI	✓/
Incision (quantitativelqualitative)	J	JI	
Undercut (quantitativelqualitative)	J	JI	✓/
Gradient (sitelmap)	J	JI	
Sinuosity (quantitativelqualitative)		JI	
Aspect (quantitativelqualitative)	J	JI	
Canopy cover (quantitativelqualitative)	/✓	JI	
Substrate size (quantitativelqualitative)	/✓	/✓	/✓
Embeddedness (quantitative/qualitative)	J	/✓	/✓
Riparian vegetation structure (quantitativelqualitative)	/✓	/✓	/✓
Fish cover (quantitativelqualitative)	/✓	/✓	/✓
Human disturbance (quantitativelqualitative)	/✓	✓/	
Discharge (quantitativelqualitative)	/✓	✓/	✓/
Floodplain width (quantitativelqualitative)	/✓	/✓	✓/
Bank erosion (quantitativelqualitative)	J		/✓
Channel type (quantitativelqualitative)	/✓		/✓
Thalweg profile (quantitativelqualitative)		✓/	
Person hours/mile (region/site)	3M‡/10	0.012M§/60	3M¶/50

* From Platts et al. (1987), Cuffney et al. (1993), Dolloff et al. (1993), Gurtz (1993), Hughes (1993), Klemm and Lazorchak (1993), Meador et al. (1993a), Meador et al. (1993b), FS (1993b), and Hayslip et al. (1994).

† Includes amphibians.

‡ Assumes 300,000 stream miles in region.

§ Assumes a random sample of 400 sites.

¶ Assumes application of methods to all stream miles.

also indicate that subjectively chosen sites can be highly unrepresentative of ecoregions. Larsen (1995) showed that 200 random sites provide the same statewide and ecoregional Index of Biological Integrity (IBI) scores as 7000 subjective sites used by Ohio EPA. Paulsen et al. (personal communication) reported that random samples provide significantly different results than subjective site selection in State, multistate, and national surveys with the probability design indicating markedly greater environmental impact in all cases.

A second reason for a randomized regional sample survey is that current monitoring hinders regional evaluations of ecosystem conditions. For example, Henjum et al. (1994) found that current data collections are inconsistent and inadequately synthesized, which precludes comprehensive status-and-trend assessments. They recommended that the Federal government establish a comprehensive, quantitative biological monitoring program for the region because of the absence of a sufficient database. They also urged that the program be founded on an appropriate sampling design for tracking ecological condition and trends.

10.4 Biological Indicators

Although there has been a perception that biological indicators are both excessively costly to sample and too variable to allow detection of status and trends, recent studies have found the opposite (EPA 1987, 1990; Plafkin et al 1989; Rhodes et al. 1994). Adler (1995) described the legal basis for biological criteria and analyses, and suggested applications for them in water programs. Biological assessments are most useful for detecting the severity of aquatic life impairments and the effectiveness of management actions. Bioindicators, bioassessments, and biocriteria are essential to the assessment of salmonid ecosystems, particularly when coupled with the use of abiotic indicators, which aid in diagnosing probable causes of deterioration or improvement (Paulsen and Linthurst 1994; Mitchell 1995). Bioindicators are especially useful and reliable when based on biomonitoring of multiple assemblages and when the data are evaluated through use of multiple metrics that incorporate both assemblage elements and processes (Fausch et al. 1990; Karr 1994; Barbour et al. 1995, Yoder and Rankin 1995). In the case of habitat conservation plans, biological monitoring, especially salmonid monitoring, also represents validation monitoring—it is the essential step in determining whether proposed habitat and management changes actually produce improvements in salmonid populations. Whatever the purpose, useful bioassessments must incorporate clear objectives, effective sampling and database management, and careful data analysis and interpretation.

Currently in the United States, three States use biocriteria in regulations and 23 States are using them in water resource management; all five States supporting natural runs of Pacific salmon are using or developing biological criteria (Southerland and Stribling 1995). In a study of the value of numerical versus narrative biological criteria at 400 stream sites, Ohio EPA found that 61% attained and 9% did not attain narrative criteria, while 34% attained and 44% did not attain numerical criteria (Yoder 1991). Clearly, increased rigor in sampling designs and methods, indicators, and analytical techniques produces greater precision, accuracy, and discriminatory power (Ohio EPA 1992).

10.5 Summary

We believe that successful salmonid conservation will be directly related to the human and fiscal resources invested in a rigorous monitoring program. Examples from wetland mitigation, forest plans, point-source discharges, and rural best management plans reveal many shortcomings in existing implementation monitoring. All four programs demonstrate insufficiently funded efforts, inadequately trained staff, unclear objectives and criteria, insufficiently used remote sensing and site visits, and lack of computerized data systems. Typically, assessment monitoring programs lack statistical designs, quantitative indicators, periodic reports and reviews, and interprogram consistency as well as the shortcomings shown by the four implementation monitoring programs. If we are to conserve salmonids and their habitats, our management actions can be treated and evaluated just as experiments are, that is, with much more rigorous design and consistent data collection at a multistate scale.