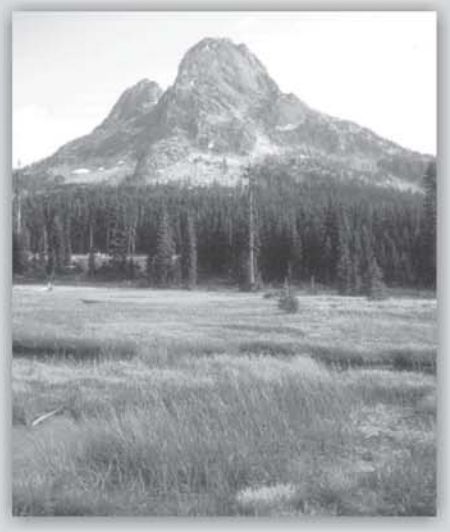
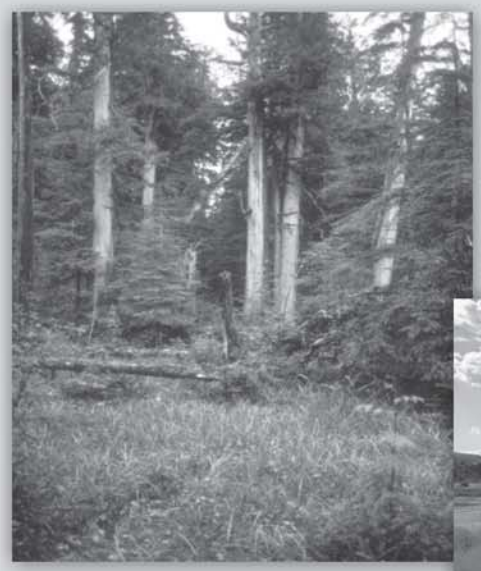
Freshwater Wetlands in Washington State

Volume 1: A Synthesis of the Science





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WASHINGTON STATE DEPARTMENT OF ECOLOGY

Washington Department of FISH and WILDLIFE

UNITED STATES ENVIRONMENTAL PROTECTION AGENCY

Freshwater Wetlands in Washington State

Volume 1: A Synthesis of the Science

DRAFT
August 2003
Ecology Publication #03-06-016

Written by: Dyanne Sheldon[1], Tom Hruby Ph.D.[2], Patricia Johnson[2], Kim Harper[1], Andy McMillan[2], Stephen Stanley[2], Erik Stockdale[2]

Edited by: Sara Noland[3] and Teri Granger[2]

1. Sheldon and Associates
2. Washington Department of Ecology
3. 2N Publications

Preferred citation:

Sheldon, D., T. Hruby, P. Johnson, K. Harper, A. McMillan, S. Stanley, E. Stockdale. August 2003 Draft. Freshwater Wetlands in Washington State Volume 1: A Synthesis of the Science. Washington State Department of Ecology Publication # 03-06-016.

This document is available online from the Department of Ecology's Website at http://www.ecy.wa.gov/biblio/0306016.html. If you require this document in alternative format, please contact Tim Schlender at 360-407-6096 (Voice) or 711 or 1-800-833-6388 (TTY). For more information, contact Teri Granger at 360-407-6857, e-mail: tgra461@ecy.wa.gov.

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Acknowledgements

Many hours of painstaking work by dedicated experts went into the production of this document. The Core Team (listed in Appendix A) persistently guided us through numerous challenges and did the bulk of the work by gathering and reading articles, dealing with problems with the database, and crafting and re-crafting the framework for the document. Katherine March also searched the public databases, generating long lists of articles to be considered for the synthesis. Kim Harper accepted the difficult challenge of managing the database.

The authors and other members of the Core Team spent countless hours reading and summarizing articles and writing and re-writing text. Of special note is Tom Hruby whose knowledge of the literature and ability to translate scientific concepts into lay terms results in his valuable technical writing skills and contribution to many chapters. Also of special note is the dedication and time spent by Dyanne Sheldon in the production of this document. She donated many hours of her own time toward the project, often burning the midnight oil to make sure articles were read and text written. Patricia Johnson deserves kudos for her work sifting through the voluminous information on compensatory mitigation and presenting it in a thorough yet easy to read manner.

Sara Noland, our chief editor, is to be complimented for her excellent editing skills. Editing this document has been a real challenge with the numerous authors involved, each with a different writing style, and the changes that were initiated relatively late in the production process. We thank her sincerely for her patience and flexibility.

Tom Noland provided invaluable assistance in scanning and converting graphics into a format suitable for publication, as well as patiently trouble-shooting computer problems.

Thanks also to Mary Lynum for her role as support staff for the project, Keren Marzi for her work on the cover, and Tim Schlender for developing the web site and posting the document on the site.

The document and this effort as a whole were the vision of Andy McMillan who also obtained the funding from several sources and pitched in to help with some of the writing.

Finally, acknowledgement must be given to Teri Granger, the project manager. Teri coordinated every aspect of the project, facilitated meetings, provided encouragement to the authors, assisted with the editing, and managed external communications while working part time.

This document would never have been produced had it not been for the financial backing provided by the U.S. Environmental Protection Agency. The Washington Departments of Ecology and Fish and Wildlife also contributed to the funding for this project. We thank the agencies for their support in undertaking such an ambitious endeavor.

Chapter 1
Introduction to the Document

This is the first part of a two-part document. This volume contains a summary and synthesis of the recent literature relevant to the science and management of freshwater wetlands. Volume 1 describes what the scientific literature says about:

- How environmental factors control the functions of wetlands across the landscape and at individual sites, how freshwater wetlands are classified according to these controls, and what functions are performed by different classes of wetlands in the state
- How human activities and land uses affect the environmental factors that control wetland functions
- How disturbances caused by human activities and land uses impact the performance of functions by freshwater wetlands
- How wetlands are protected and managed using common tools such as buffers and compensatory mitigation
- How cumulative effects result from the current use of these tools

The second volume will translate these scientific findings into options and recommendations for policies and regulations that can be used to protect and manage wetlands.

This work is being collectively prepared by the Washington State Department of Ecology (Ecology), the Washington State Department of Fish and Wildlife (WDFW), and a private consultant. The U.S. Environmental Protection Agency (EPA) funded and assisted in its production. Representatives from these agencies, as well as staff from the private consultant, made up a team (the Core Team) that guided the project. See Appendix A for a list of Core Team members.

Both volumes will be of use to all those interested in protecting and managing wetlands. They will find these documents useful in gaining a greater understanding about the current science of the ecology, function, and management of freshwater wetlands. Examples of user groups include federal, state, and tribal staff; planners; resource managers; wetland scientists; builders; farmers; environmentalists; and other concerned citizens.

One key group involved in wetland protection in the State of Washington is local governments. Through the Growth Management Act (RCW 36.70A) every county and city in Washington must identify and inventory critical areas (including wetlands) within their boundaries and protect them. In 1995, an amendment to GMA

(RCW 36.70A.172[1]) required that all city and county governments rely on best available science (BAS) when developing their critical areas policies and regulations.

WAC 365-195-900 provides assessment criteria to assist in determining whether information constitutes the best available science. To be best available science, scientific information must be produced through a valid scientific process. A valid scientific process is one that produces reliable information that is useful in understanding the consequences of regulatory decisions and in developing policies and regulations that will be effective in protecting the functions and values of wetlands and other critical areas.

Appropriate sources of scientific information as defined in WAC 365-195-900 include:

- Research
- Monitoring
- Inventory
- Survey
- Modeling
- Assessment
- Synthesis
- Expert opinion

Information derived from one of these sources can be considered scientific information if it possesses the required characteristics shown in WAC 365-195-900. See Appendix B for characteristics of a valid scientific process and a description of the common sources of scientific information as listed and defined in WAC 365-195-900.

Volume 1 meets the definition and characteristics required for a synthesis. Findings from scientific journal articles, government publications, technical books, and other sources that meet the definition and characteristics of BAS in WAC 365-195-900 were used and referenced in the synthesis. Conference proceedings, personal communications, and other "gray" literature were occasionally used and in some cases had not been peer reviewed. Reviewers of Volume 1 are asked to judge the reliability of the sources we used, including any gray literature.

In some cases we have cited unpublished data collected during the calibration of the Washington State Wetland Function Assessment Methods and the Washington State Wetland Rating System. These data have not been published in scientific journals. However, the assessment methods and the wetland rating system, which the data support, have been peer reviewed.

This synthesis, which is undergoing broad peer review, may be of special interest to local governments that do not have the resources to complete their own review of the scientific literature. Local governments should, however, also consider locally and regionally specific information not included in this synthesis if such information meets the

characteristics of a valid scientific process and is one of the specified sources of scientific information described in Appendix B.

1.1 Scope of Volume 1

The scope of this project, and the information in the synthesis, is limited to freshwater wetlands of Washington State. It does not address streams or riparian areas that are not wetlands. We do however summarize literature related to stream buffers.

Marine and estuarine systems were excluded primarily to keep the scope of the project in the range of the available funding. Some recent scientific information on coastal and estuarine wetlands has been summarized by WDFW, Ecology, and other agencies through the Aquatic Habitat Guidelines Project, which is available on the internet (www.wa.gov/wdfw/hab/ahg).

The effects of growing cranberries is wetlands are also not covered in this document because of the time and funding constraints of the project and the limited area of the state that is affected by cranberry production. In addition, information related to the effects of silviculture and forest practices on forested wetlands is not included because this subject is being addressed in another document currently being developed (Cooke in press).

In addition, the synthesis in Volume 1 is limited to information that has a practical application to the management and protection of freshwater wetlands. For the most part, available documents from the past 10 years were used as the primary sources for this report. It was assumed that this most recent literature would incorporate relevant science from the preceding years. Older documents were used in instances where they had not been superseded by more recent studies.

This volume DOES NOT contain agency recommendations or suggestions for implementation. Any recommendations provided in Volume 1 (for instance, in the section of Chapters 5 and 6 addressing buffers and compensatory mitigation) are those that have been described in the literature and are included here only as part of the synthesis of existing scientific information.

1.2 Overview of Volume 1

Volume 1 is organized into seven chapters. The chapters share a common organization, beginning with a reader's guide that describes the topics covered in the chapter and how the chapter is organized. An introduction then provides general background information, definitions, and clarifications. Each chapter describes how well the subject is documented in the literature, particularly for the Pacific Northwest. The chapters note data gaps where adequate or appropriate literature on an issue could not be found. Key

points are summarized at the end of major sections and conclusions provided at the end of each chapter.

A brief summary of the contents of each chapter and the appendices follows. In the synthesis page numbers are assigned to each chapter individually and are not sequential. For example, there are seven page number 3s, one for each chapter (1-3, 2-3, 3-3, etc).

Chapter 1 – Introduction to the Document

Chapter 1 describes the purpose and background for this synthesis and the many groups that may find the information useful. It describes how it relates to requirements under the Growth Management Act that local governments include "best available science" when developing and revising ordinances for critical areas including wetlands. It gives a brief synopsis of the intent and summarizes the organization of Volume 1. The methods used to prepare and review Volume 1 are provided in Appendix C at the end of the document.

Chapter 2 – Wetlands in Washington and How They Function

Chapter 2 summarizes the science regarding how wetlands function in Washington State. It describes how functions are defined and introduces the concept that the performance of functions is controlled by a number of environmental factors within the wetland boundary (site scale) as well as in the broader landscape (landscape scale). The chapter then describes how some of the key factors that control functions are used to classify wetlands into groups that perform functions in similar ways.

The chapter goes on to describe 15 functions of freshwater wetlands in Washington. Where applicable, the chapter discusses the differences in functions among wetland classes and in various areas of the state. The major functions described are those that were defined for the Washington State Function Assessment Methods (Hruby et al. 1999, 2000).

Chapter 3 – Environmental Disturbances Caused by Different Human Activities and Uses of the Land

In Chapter 3, the discussion shifts from wetland functions and the environmental factors that control the performance of functions to the major disturbances created by human activities and uses of the land and water. In this context, a *disturbance* is an event that changes an environmental factor that controls wetland functions. Ten disturbances are identified such as changing the amount of sediment and increasing the concentration of salt. A description of the movement of water on the landscape and thresholds at which impervious surfaces impact aquatic resources, general disturbances to the quality of water, and general disturbances to connections between habitats is provided.

The chapter continues with separate sections for four major types of human land uses in Washington State (agriculture, urbanization, logging, and mining) and how they create disturbances. For each of these four land uses, several types of disturbances that change the factors controlling wetland functions are discussed, including (as applicable):

- Changing the physical structure within a wetland (e.g., filling, removing vegetation, tilling soils, compacting soils)
- Changing the amount of water in wetlands (increasing or decreasing the amount)
- Changing the fluctuation of water levels (frequency, amplitude, direction of flow)
- Changing the amount of sediment (increasing or decreasing the amount)
- Increasing the amount of nutrients
- Increasing the amount of toxic contaminants
- Changing the acidity (acidification)
- Increasing the concentration of salt (salinization)
- Decreasing the connection between habitats (fragmentation)
- Other disturbances (noise, etc.)

Chapter 4 – Impacts of Human Disturbances on the Functions of Wetlands

Chapter 4 integrates the concepts discussed in Chapter 2 and Chapter 3. Chapter 2 described the functions performed by wetlands and the environmental factors that control these functions. Chapter 3 discussed the major disturbances caused by different human activities and uses of the land. Ten sections, one for each of the disturbances listed above, summarize how these disturbances ultimately leads to impacts on hydrologic functions, functions that improve water quality, and habitat functions.

Chapter 5 – The Science and Effectiveness of Wetland Management Tools

Chapter 5 presents a synthesis of what the current literature reports on four tools currently used to address impacts to wetlands and their functions: wetland definitions, wetland delineation methods, wetland ratings, and buffers. In the section on definitions, biological versus regulated wetlands (including "prior converted"), small wetlands, and those that are isolated are discussed. This chapter does not provide language or recommendations for regulatory or policy language—those will be provided in a separate volume on management options and recommendations (Volume 2).

Chapter 6 – The Science and Effectiveness of Wetland Mitigation

Chapter 6 discusses another commonly used wetland management tool, compensatory wetland mitigation. This topic is discussed in its own chapter because of the large volume of literature available on this subject. Topics covered in this chapter include:

- Evaluation of the success of compensatory mitigation
- Compliance of mitigation projects with permit requirements

- Types of compensatory mitigation
- Replacement ratios and replacement of wetland acreage
- Functions provided by compensatory mitigation projects
- Reproducibility of particular types of wetlands (bogs, fens, vernal pools, alkali wetlands, and mature forested wetlands)
- Suggestions from the literature for improving compensatory mitigation

Chapter 7 - Cumulative Impacts on Wetlands

Chapter 7 describes how the current wetland management approach of protecting resources parcel by parcel can lead to inadequate protection of wetlands. It discusses some reasons given by researchers for the difficulty in assessing cumulative impacts.

The chapter summarizes the recommendations by numerous researchers for ways to manage aquatic resources on a landscape level rather than case by case. Watershed planning and zoning, assessment and selection of restoration sites, floodplain restoration, and other topics are discussed.

Glossary

The glossary provides definitions of technical terms used throughout Volume 1.

References

The references cited in the text are listed in a separate section at the end of Volume 1. Some of these references represent overviews in which a researcher describes trends observed from numerous studies conducted in previous years; in these cases, the reference list includes only the overview document and not all of the previous studies (unless those previous studies were actually reviewed by the Core Team). Similarly, where Volume 1 quotes a literature source and the quotation includes a citation to another author's work, that citation is included in the reference list only if the Core Team actually reviewed it.

Portions of Adamus et al. (2001), a review of current scientific literature on the impacts of human activities on wetlands and their functions, were adapted where relevant into Chapter 4 with permission from Dr. Adamus. The list of cited references at the end of the document includes the literature sources from those portions of Adamus et al. (2001) that were adapted.

The authors are aware that the list of cited references is not consistently formatted and abbreviations are used. This has resulted from various problems with the database used to generate the list. Correcting these problems with the database requires significant time and effort. The Core Team decided to publish the draft with these imperfections, while striving to include all citations made in the text of the document.

Appendices

The appendices of Volume 1 are as follows:

- Appendix A identifies the team guiding the production of Volume 1 (the Core Team).
- Appendix B describes the characteristics of a valid scientific process and types of scientific information defined by the Washington Administrative Code.
- Appendix C details the methods used in the literature review and production of Volume 1.
- Appendix D is a placeholder where those who comment on this draft of Volume 1 will be listed in the final version of this document.
- Appendix E provides information about various terms and methods that have been used to organize and group information about wetlands, such as classification, characterization, rating, etc.

1.3 Methods for Preparing and Reviewing Volume 1

The primary steps taken to arrive at publication of this document include:

1. Searching the literature
2. Reviewing, sorting, and prioritizing the reference lists
3. Obtaining the reference documents
4. Reading and entering notes on the documents in a database
5. Writing and revising the text
6. Obtaining peer and public review
7. Responding to comments, revising the text, and completing the document

With the distribution of this draft, step 6 is currently underway.

Given the scope of this project and the volume of existing information on wetlands, each of these steps has entailed a lengthy process. The processes used for these steps, including the scientific databases and the key words used to search them, are described in Appendix C.

1.4 Public Involvement Process for Volume 1

1.4.1 Public Involvement Process

The public involvement process for Volume 1 included several mailings and focus group meetings. Ecology compiled a mailing list of experts, local government planners, and other groups and individuals from various existing mailing lists compiled for other wetland-related projects.

In October 2001, Ecology sent out a focus sheet describing the project and a cover sheet that solicited the recipient's participation in the project. This sheet included a tear-off card that could be used to request that the sender be retained on the mailing list. The mailing list was then edited based on the returned cards.

Focus group meetings were held in January 2002 in Moses Lake and Olympia to begin to gather public input for the project. These meetings were attended by various Core Team members, local planners, other local government staff, and other interested parties. The purpose of these meetings was to help focus the project so that the synthesis reports meet the needs of local governments. The meetings gave opportunities to the Core Team to present information on the project and to listen to questions and concerns from the attendees. Keyword lists were revised based on focus group input.

In June 2002, Ecology sent out a mailer with a project update to the entire mailing list, discussing the status of the project, changes in the timeline, changes in deadlines for critical area ordinance revisions, and other issues.

In November 2002, Ecology staff contacted selected experts in various disciplines to solicit their review. The list of peer reviewers was not intended to be inclusive of all experts. The purpose was to make sure that each of the major topics in Volume 1 was reviewed by one or more recognized experts in that discipline.

In February 2003, Ecology sent another mailing to all those on the list to determine who wanted to comment on the draft of Volume 1. In June 2003, Ecology distributed a notice via email to update the public on revised target dates for distribution of the draft document for peer and public review.

1.4.2 Current Public Review Process

The draft of Volume 1 has been posted on the Ecology web site and reviewers are encouraged to download the document. The draft was also mailed directly to those who requested a CD or paper copy. Experts are reviewing the document at the same time as local government and agency staff, interested organizations, and the public.

Commentors are asked to complete a questionnaire and submit detailed comments six weeks after it has been distributed. A deadline for submitting comments is provided in instructions that accompany this document. Ecology staff will collect the questionnaire

and written comments, organize them, and present them to the Core Team for discussion. Authors will revise the text of Volume 1 accordingly.

Volume 1 is available online

Ecology has developed a home page for this project on the Shorelands and Environmental Assistance Program web site. The home page includes a project description, contact information, current status of the project, and copies of all the updates that were sent. The "current status" section of the home page is updated periodically. The web page also includes a copy of this draft Volume 1. See the instructions for this document for the address of the web site.

1.5 Conclusions

Volume 1 provides a summary of relevant scientific literature related to freshwater wetlands in the Pacific Northwest and their management. The document will be useful to all those who have an interest in the management and protection of wetlands including agency staff, consultants, interested organizations, and citizens. It will be particularly helpful to local governments who are required under the Growth Management to include best available science when developing and revising regulations protecting critical areas including wetlands. Volume 1 will be reviewed by technical experts (peer reviewed) and any other interested parties. The intention of the project and the review process is to produce a synthesis of the current science on freshwater wetlands that is easily understood, yet thorough and scientifically rigorous.

Chapter 2
Wetlands in Washington and How They Function

2.1 Reader's Guide to This Chapter

Chapter 2 presents information on wetlands in Washington and how they function. It introduces the concept that the performance of functions is controlled by a number of environmental factors within the wetland boundary (site scale) as well as in the broader landscape (landscape scale). The chapter then describes these controls of function relative to regions and wetland types (classification of wetlands) in Washington before turning to detailed descriptions of the functions performed by the wetlands east and west of the Cascade Mountains and in different wetland classes.

To manage and protect wetlands, an understanding of wetland functions must be supplemented by knowledge of how these functions are affected by human activities. Chapter 3, therefore, goes on to describe how various land uses and activities disturb the environment, for example by causing excess nutrients, increased runoff and fluctuating water levels, and disconnected habitat. These disturbances in turn affect the environmental factors that control wetland functions. Chapter 3 describes what the literature says about the disturbances created by different land uses, while Chapter 4 goes into detail regarding how each disturbance affects particular wetland functions, including the organisms that use wetlands.

2.1.1 Chapter Contents

Major sections of this chapter and the topics they cover include:

Section 2.2, Introduction and Background on Wetland Functions defines the term "wetland functions" and explains how the concept of wetland functions has evolved over the last few decades. The section describes how ecological processes at both large and small scales control the functions provided by wetlands. Regional differences in wetland functions are summarized. The difference between "functions" and "values" is explained.

Section 2.3, Classification of Wetlands in Washington as a Key to Understanding their Functions describes the ecological regions in Washington State and how wetlands across the state are classified into like-functioning groups. The classes and subclasses of wetlands found in the state are described.

Section 2.4, Overview of Wetland Functions in Washington State introduces the functions of wetlands that are currently the focus of management efforts. These functions

fall into three main categories: water quality improvement, hydrologic functions, and the provision of habitat.

Section 2.5, How Wetlands Perform Functions in Washington State details each of the wetland functions listed in Section 2.4. For each function, the text provides a general description of how the function is performed, then goes into detail about how that function is performed by wetlands of various classes and in different areas of Washington.

Section 2.6, Chapter Summary and Conclusions ties together the major concepts presented in the chapter.

2.1.2 Where to Find Summary Information and Conclusions

Each major section of this chapter concludes with a brief summary of the major points resulting from the literature review on that topic in a bullet list format. The reader is encouraged to remember that a review of the entire section preceding the summary is necessary for an in-depth understanding of the topic.

For summaries of the information presented in this chapter, see the following sections:

- Section 2.2.4
- Section 2.3.6
- Section 2.4.4
- Section 2.5.4

In addition, Section 2.6 provides a summary and conclusions about the overarching themes gleaned from the literature and presented in this chapter.

2.1.3 Data Sources and Data Gaps

Much of what we know about wetlands in Washington and how they function is based on the collective expertise and judgment of teams of experts who developed the Washington State Wetland Function Assessment Methods and who are revising the Washington State Wetland Rating System. Both tools are methods that analyze the functions of wetlands in the state. This expert, regional information is critical because much of the knowledge about wetland functions was developed outside the Pacific Northwest. The assessment methods and documents on the revised rating system can be considered a synthesis of the best available science for understanding the functions performed by Washington's wetlands. The wetland scientists who developed these reports analyzed existing information and extracted material that is relevant for Washington State as well as adding their best professional experience, expertise, and judgment.

These efforts were based on a formal process that developed a consensus among wetland scientists in the region. This process has included peer review and public comment. The documents resulting from the function assessment project and the rating system effort are frequently cited in this synthesis as Hruby et al. (1999), Hruby et al. (2000), and Hruby (2003). More information about these efforts is provided in Section 2.3.2. Information is also available at http://www.ecy.wa.gov/programs/sea/wetlan.html.

Major gaps in our knowledge of how wetlands in Washington function, however, still exist for the types of wetlands for which assessment methods have not yet been developed. For example, there is little published information about the functions of "slope" wetlands and "flats" wetlands. There is also less literature on the wetlands in the arid region of the state.

2.2 Introduction and Background on Wetland Functions

2.2.1 An Evolving Understanding of Wetland Functions

The concept of wetland functions, the things that wetlands "do" (Brinson et al. 1995), is relatively new in both the regulatory and scientific arenas. For many years wetlands were considered nuisances and wastelands (Department of Natural Resources 1998). The ecosystem found within a wetland was not considered important enough to study and understand. Today, however, we know that the ecosystem in wetlands is important and interacts with other aspects of the landscape around it. We have found that the structural components of a wetland and its surrounding landscape (such as plants, soils, rocks, water, and animals) interact with a variety of physical, chemical, and biological processes to perform functions.

The concept of wetland functions has evolved since it was first introduced about four decades ago. Wetlands were first considered primarily to function as habitat for important species such as waterfowl. The factors that were thought to control how a wetland functions in this respect were the structural elements in a wetland. For example, how much open water did the wetland contain? What types of vegetation were found there? This interest in wetland structure led to the development of a classification system for wetlands in 1979 based on the vegetation and water regime (Cowardin et al. 1979). This system is still in use today. See Section 2.3.1 for more on this classification system.

It soon became apparent, however, that wetlands contribute more to the ecosystem in a watershed than just habitat. During the 1980s much research was done on how wetlands filter pollutants and improve water quality in a watershed. As a result, wetland engineers started to design and create wetlands specifically to treat wastewater (Hammer 1988). In this decade, wetlands were also recognized for their contribution to flood protection (Adamus et al. 1987).

The ongoing research in the 1980s also led to a realization that the functions performed within a wetland are controlled by a number of environmental factors both within and

outside of the wetland. Climate was recognized as the major factor that affects how wetlands function at the largest geographic scale (Bailey 1995, Benda et al. 1998). Differences in temperature, rainfall, and seasonal and annual changes impact all aspects of natural ecosystems, including wetlands.

During the 1990s Brinson (1993a) and the National Academy of Sciences (National Research Council 1995) described and defined three other factors at a smaller geographic scale that can be considered primary controls of functions within a wetland:

- Geomorphic or topographic setting of the wetland
- Direct source of water to the wetland
- Hydrodynamics, or the direction of flow and strength of water movement within the wetland

More recently, however, scientists have become increasingly aware that functions performed by wetlands are also controlled by processes that occur at the scale of the watershed. There is currently an emphasis on trying to understand wetland functions in the context of how water, sediments, and nutrients move in a watershed (Bedford 1999). The surface geology and soils, the routing of water through the watershed, and the movement of sediments, large wood, nutrients, and other chemicals are all considered important factors in controlling how individual wetlands function (see Section 2.2.3).

2.2.2 How Wetland Functions Are Defined

The things that wetlands "do" (that is, their functions) occur at many scales as well, from the microscopic (such as bacterial decomposition of organic matter) to the continental (such as providing refuge and feeding for migrating waterfowl along the continental flyways). If every thing that occurs within a wetland were identified as a separate function, the number of functions would be almost infinite. For example, the decomposition of organic matter by bacteria is a combination of many types of decomposition, one for each individual species of bacteria found in the wetland. Each bacterial species decomposes organic matter at a different rate and under different environmental conditions. Each of these could be considered a separate wetland function.

In contrast, a function can be a broad "lumping" of many environmental processes. For example, the "removal of imported elements and compounds" is a function identified in one method for assessing wetland functions (Brinson et al. 1995). At least a dozen nutrients and several hundred known contaminants can be found in surface waters. Therefore this function combines several hundred different removal processes, one for each imported nutrient, contaminant, and other compound.

Furthermore, wetlands perform many types of functions, but not all wetlands perform the same functions, nor do similar wetlands provide the same functions to the same level of performance (Clairain 2002).

One of the initial tasks in defining functions, therefore, is to identify and group the processes and interactions that occur in wetlands into some manageable number of "functions." Most functions are generally grouped in terms of three broad categories (Adamus et al. 1991):

- Biogeochemical functions, which are related to trapping and transforming chemicals and include functions that improve water quality in the watershed
- Hydrologic functions, which are related to maintaining the water regime in a watershed and include such functions as reducing flooding
- Food web and habitat functions

Functions are subdivided into more specific groups by the environmental processes or interactions within the wetland that are related and are on a similar temporal and spatial scale. They are also grouped based on the needs for managing wetlands (Hruby 1999). For example, managers may need to know how well a wetland removes specific constituents that contribute to poor water quality such as sediment, nutrients, and toxic compounds, rather than having only a general assessment of the removal of imported elements and compounds.

Table 2-1 gives examples of how the many different processes and interactions that occur in wetlands have been grouped under different names for various policy and regulatory purposes. They are organized into the three broad categories above (water quality improvement, hydrologic functions, and food webs and habitat).

The names of the categories to some degree reflect how broadly the function is defined. "The removal of all imported elements and compounds" is a broadly defined function, whereas "removing sediment" is a more narrowly defined function. Section 2.4 describes in more detail the functions that have been chosen for the Washington State Wetland Function Assessment Project and the Washington State Wetland Rating System.

Table 2-1. Different ways of dividing wetland functions.

Wetland Evaluation Technique (WET) [a]	HGM Guidebook for Riverine Wetlands [b]	Mill Creek Special Area Management Plan[c]	WA Functions Assessment Method – Lowlands of Western Washington [d]
Biogeochemical Functions Related to Improving Water Quality			
Nutrient Removal/ Transformation	Nutrient Cycling	Nutrient Uptake	Removing Nutrients
Sediment Stabilization	Removal of Imported Elements and Compounds	Sediment Stabilization	Removing Sediment
Sediment/Toxicant Retention	Retention of Particulates	Retention of Toxics	Removing Metals and Toxic Organic Compounds
Hydrologic Functions Related to Maintaining the Water Regime			
Floodflow Alteration	Dynamic Surface Water Storage	Floodflow Alteration	Reducing Peak Flows
Groundwater Recharge	Long-term Surface Water Storage	Groundwater Discharge	Decreasing Downstream Erosion
Groundwater Discharge	Energy Dissipation		Recharging Groundwater
	Subsurface Storage of Water		
	Moderation of Groundwater Flow or Discharge		
Functions Related to Maintaining Food Webs and Habitat			
Aquatic Diversity/ Abundance	Maintain Spatial Structure of Habitat	Habitat for Aquatic Species	General Habitat
Wildlife Diversity/ Abundance/ Migration Wintering	Maintain Interspersion and Connectivity	Habitat for Anadromous Fish	Habitat for Invertebrates
Production Export	Maintain Distribution and Abundance of Invertebrates	Habitat for Resident Fish	Habitat for Amphibians
	Maintain Distribution and Abundance of Vertebrates	Habitat for Migratory Birds	Habitat for Anadromous Fish
		Habitat for Resident Birds	Habitat for Resident Fish
		Habitat for Other Species	Habitat for Wetland-Associated Birds
			Habitat for Wetland-Associated Mammals

Sources:
[a] Adamus et al. (1987)
[b] Brinson et al. (1995)
[c] U.S. Army Corps of Engineers (2000)
[d] Hruby et al. (1999)

Relationship of functions to values

The scientific literature has in the past confused the terms wetland "functions" and wetland "values." In fact, the term "functional values" was in common usage during the 1980s and early 1990s (e.g., Amman et al. 1986). The correct interpretation of the term "functional values" suggests that wetland values were functioning, which was not the intent of the phrase. As mentioned previously, wetland functions are the environmental processes that take place in a wetland. Society, however, does not necessarily attach value to all functions. Value is usually associated with goods and services that society recognizes, and not all environmental processes are recognized or valued.

The National Research Council (1995) says the following about the differences between values and functions.

> *Because value is a societal perception, it often changes over time, even if wetland functions are constant. Value can change over time as economic development changes a region. The value of a wetland in maintaining water quality near a source of drinking water can be great even if the wetland is small (Kusler 1994). Some values can be mutually exclusive if they involve direct or indirect manipulation, exploitation, or management of wetlands. For example, production of fish for human consumption could conflict with the use of a wetland to improve water quality of water that contains toxins.*

Sometimes the values provided by wetlands are called functions because they are based on some aspect of the wetland ecosystem rather than a product of it. For example, "recreation" is a wetland value that is often defined as a function (Adamus et al. 1987) even though it is not something a wetland "does." Other values that have been called functions include "education" and "aesthetic quality" (Roth et al. 1993). These are important values of wetlands that cannot, however, be assessed or rated using the same methods used to assess functions based on wetland processes (Hruby 1999).

2.2.3 Environmental Factors that Control Wetland Functions

Functions of wetlands, as defined previously, represent interactions among the different components of the ecosystem and the landscape. Thus, functions can be influenced or controlled by changes to any one of these components. For example, a wetland may perform the function of providing overwintering habitat for coho, for which the presence of surface water is critical. This function will, therefore, change if the wetland is drained so no surface water remains. Changes in functions, however, can also be a result of alterations to the watershed outside the wetland boundary. For example, surface water in the wetland may also be eliminated if its water supply is diverted. Also if the gravel beds

in which the coho spawn farther up in the watershed are disturbed, or if the flow in the stream is reduced to such an extent that the young can no longer swim to the wetland from the spawning areas, the wetland's support of coho overwintering habitat will be altered.

Likewise, the expression of one function in a wetland (such as habitat) can result in a change to the larger scale ecosystem and the landscape. For example, if the conditions are right for beavers to settle in a wetland along a stream or river (i.e., the wetland functions as good habitat), the beavers will build a dam and create a ponded wetland. This will change the vegetation in the wetland and possibly alter other wetland functions such as improving water quality and storing flood waters. These changes may be significant enough to change the water quality and the movement of water through that part of the watershed (a change in one of the primary controls of function).

Any factor that changes how well, or how much, a function is performed by a wetland can be considered a "control" of that function. Another term often used in the scientific literature is "driver." The "drivers" of functions in wetlands determine how well the functions are performed. An action or occurrence that affects a "control" or "driver" is called a "disturbance" by ecologists (Dale et al. 2000, Carpenter et al. 2001). The type, intensity, and duration of disturbances can change the physical structure of the ecosystems and how they behave (ecosystem dynamics) (Dale et al. 2000).

Human activities create a disturbance that causes a "stress" on the ecosystem to which it responds. Scientists often use the term "stressor" to distinguish those disturbances that have a significant impact on an ecosystem from those that have little impact (see for example Adamus et al. 2001, Laursen et al. 2002).

In this report, however, we are not using the term "stressor." All the disturbances discussed and reviewed here are significant enough to impact wetlands and their functions. To avoid confusion, the term "disturbance" is used throughout this document.

Human uses of the land create a different set of disturbances than were present before human activities modified the land (Dale et al. 2000). The disturbances that are caused by human activities are discussed in Chapter 3, and the impacts these disturbances have on wetlands and their functions are described in Chapter 4.

The focus of research and management has been on functions and controls of functions that occur within the wetland itself and less on those that are a part of the ecosystem and landscape of the entire watershed. This has resulted from the fact that the need to define wetland functions and determine their controls has actually been driven by regulatory requirements and policy (Brinson et al. 1995, Clarain 2002). The policy has been to have a "no net loss of wetland area and function" at both the state and the national levels. However, this focus on functions and their controls being confined to the wetland itself is changing. We are learning that managing wetlands requires an understanding of the

"relationship of the individual wetlands to the landscape" (Bedford 1996) as well as the wetland itself.

A summary of the literature addressing the environmental factors that control wetland functions is presented below. First reviewed is the literature that addresses controls that occur at the scale of the wetland's contributing basin (that part of the landscape that contributes surface water to the wetland). The controls that are found within the boundary of the wetland (the site scale) are then described. The discussion includes a number of conceptual models that have been developed to help visualize and understand the complex interactions between wetland functions and environmental factors at different scales.

Terms used in this document to refer to drainage systems

Surface and subsurface water flows through the landscape within *drainage systems*. These drainage systems are often called basins, sub-basins, watersheds, or river basins depending on the size of the area. In this document, drainage systems are generally referred to using one of two terms:

- *Watersheds:* A watershed is a geographic area of land bounded by topographic high points in which water drains to a common destination.
- *Contributing basins:* A contributing basin is an area from which water drains to a particular wetland.

2.2.3.1 Controls of Functions at the Scale of the Contributing Basin to the Wetland

Climate, geology, and the hydrologic characteristics in a watershed control how water, sediment, and nutrients move (Bedford 1999). Together, along with factors within the boundary of a wetland, these factors control the functions performed. Scientists call these large-scale environmental factors the *hydrogeologic setting* of a wetland (Winter 1983, 1986, 1988, 1989, 1992, LaBaugh et al. 1987, Winter and Woo 1990). The following describes some models that have been developed to better understand these controls of wetland functions.

A "hydrogeologic" model created by Bedford (1996, 1999) concludes that wetlands develop and persist over time through the interaction of the hydrologic cycle with the landscape (Figure 2-1). This model views wetlands as part of an ecological system that is continuous with large-scale surface and groundwater systems. In this model, several geologic characteristics control the flow and chemistry of water, including the surface relief and slope of the land, the thickness and permeability of the soils, and the composition and hydraulic properties of the underlying geologic materials (Bedford 1999).

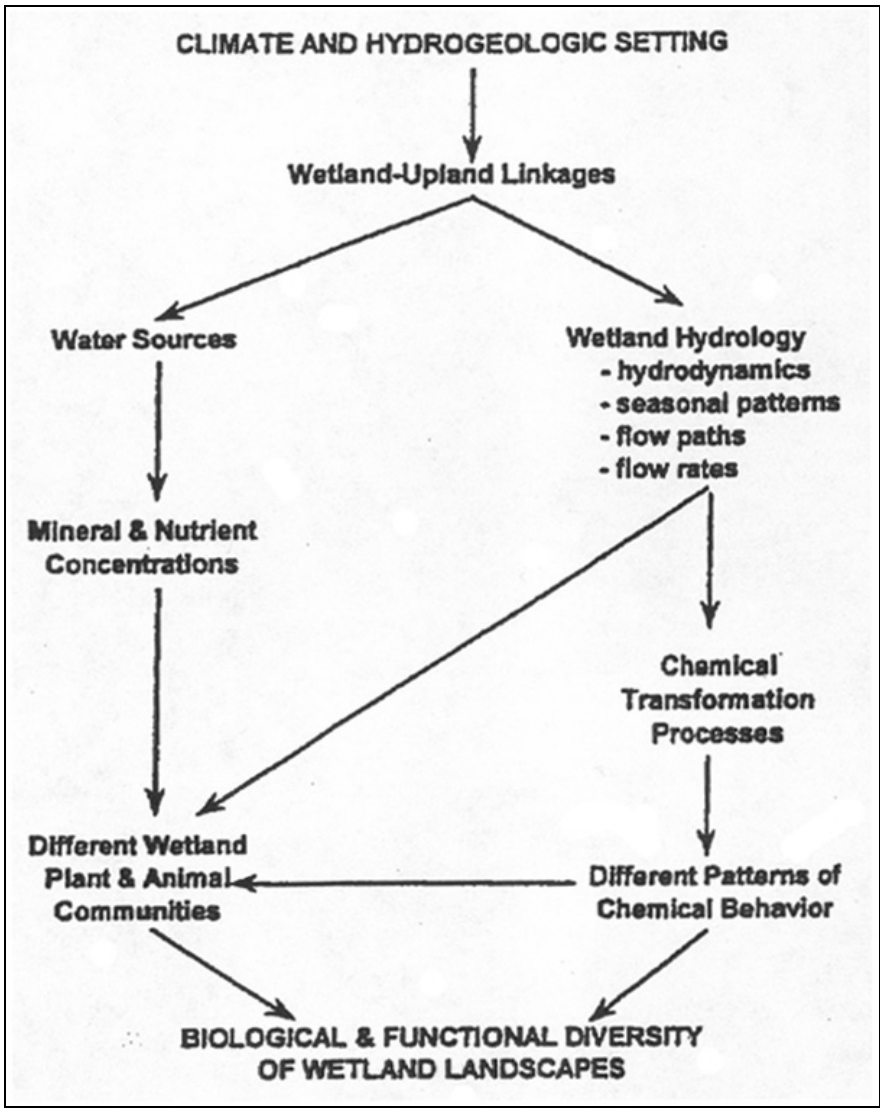

Figure 2-1. A model of the environmental factors that control wetland functions (Bedford 1999; reprinted with permission).

In Bedford's hydrogeologic model, as in all the models discussed here, climate drives the large-scale water regime. Climate determines the precipitation and patterns of evapotranspiration that ultimately move surface and groundwater into and out of wetlands (see Figure 2-1). It also determines how sediments and chemicals (e.g., salts and nutrients) are eroded from bedrock and transported throughout the system.

A similar model to that of Bedford considers the contributing basin of a wetland in describing the factors that affect functions. This model, known as the "process-structure-function" model (Figure 2-2), was developed in conjunction with restoration plans for Northwest riverine systems. It is described in more detail in Beechee and Bolton (1999), Gersib (2001), and Stanley and Grigsby (2003). The model assumes that the biological, physical, and chemical characteristics (structure and functions) of aquatic systems including wetlands are determined by the interaction of many processes operating at the

larger scale of the landscape (Kaufman et al. 1997, Beechie and Bolton 1999). These processes include the movement of (Naiman et al. 1992):

- Water (surface and subsurface)
- Sediment
- Nutrients and other chemicals (salts, toxic contaminants)
- Large woody debris
- Energy (in the form of sunlight)

According to the "process-structure-function" model, the interactions of these processes with climate and geomorphology determine the structure within wetlands (e.g., substrate, plant species). The wetland structure, in turn, is one factor that influences the type and performance of wetland functions.

For example, a wetland may produce large quantities of plant material and support the function of a rich food web. In order to provide this function, the wetland needs to have waters rich in nutrients coming into it, good exposure to sunlight, and a way for the production of plant material to leave the wetland into surrounding aquatic ecosystems. The major controls for this function are the movement of water to and from the wetland, the movement of nutrients into and within the wetland, and an adequate source of energy.

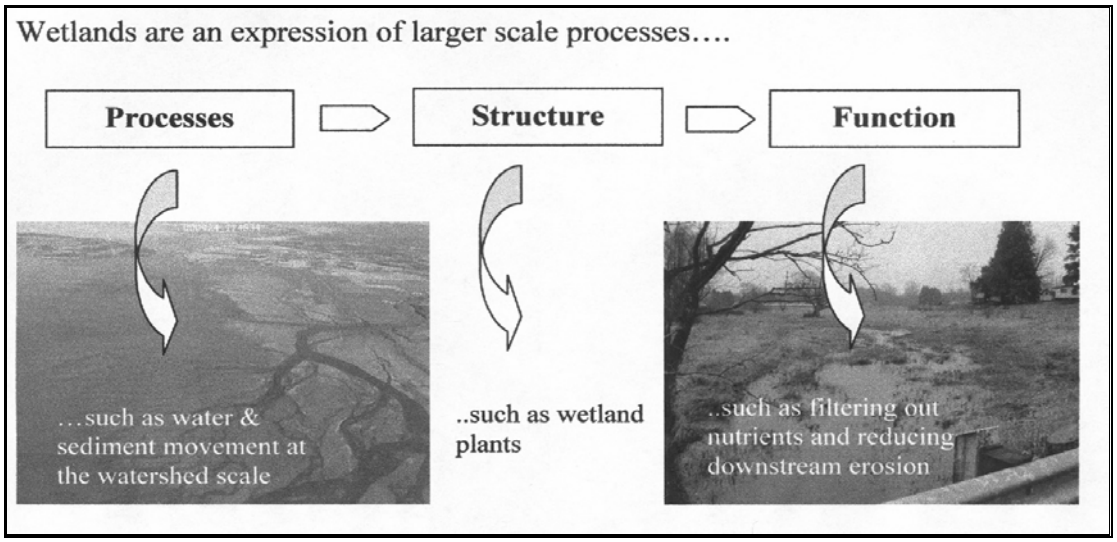

Figure 2-2. Process-structure-function model.

The "process-structure-function" model, like Bedford's, assumes that changes in land use affect processes such as the delivery of water, nutrients, sediment, and toxics to aquatic systems (Poiani et al. 1996, Mallin et al. 2000). These in turn affect structure and function within those aquatic systems.

2.2.3.2 Controls of Functions at the Site of the Wetland

The environmental factors at the large scale that control functions in a wetland ultimately affect the environmental factors within the wetland itself (the site scale). Brinson (1993a) has developed a model that defines three factors at the site scale that can be considered as primary controls of wetland functions (see the list below). Brinson's (1993a) model also uses characteristics of the landscape as factors that control functions in a wetland, but his model focuses primarily on the wetland itself relative to the two models discussed earlier (Bedford 1999). For example, Brinson's model emphasizes the shape and location of the wetland in the landscape and the type of water movement in the wetland that is dominant. In contrast, the "hydrogeologic" and process-structure-function models describe the surface and subsurface conditions across the landscape that control water processes within the wetland's contributing basin.

- The **geomorphic setting** (landscape position) of the wetland. Geomorphic setting is the topographic location of the wetland within the surrounding landscape and the geology that underlies it. In other words, is the wetland in a depression, on a slope, in a floodplain, or on the shores of a lake? The underlying geology also determines the soils present in the wetland, and this for example has an effect on the type and abundance of the plants found there.

- The **source of water** to the wetland. The sources of water can be simplified to precipitation, surface flow, shallow subsurface flow, and groundwater.

- The **hydrodynamics** of the wetland (the direction of flow and strength of water movement). Hydrodynamics refers to the movement of water in the wetland and its capacity to do work. There are three qualitative categories of hydrodynamics: (1) vertical fluctuations of the water levels or water table, (2) unidirectional surface or near-surface flows that range from strong currents contained in channels to slow sheet flow down a slope, and (3) bidirectional flows resulting from tides or wind-driven currents in lakes.

The Brinson model (1993a) is the basis of the hydrogeomorphic (HGM) classification system which clusters wetlands into similarly functioning groups. The classification system, and an earlier classification used for habitat mapping, are described in Section 2.3.1.

2.2.3.3 Summary of the Controls of Wetland Functions

The different models described above that have been developed to help understand how wetlands function are the basis for the following chapters that describe the impacts of human activities on wetlands and their functions. Unfortunately, the many articles that have been written on the subject of wetland functions and how they are controlled by environmental factors have engendered some confusion in the terms used. For example, the term "process" has been used by different authors to describe a wide range of happenings that include the routing of water at a landscape scale as well as the chemical

reactions by which bacteria change nitrate to nitrogen gas at the microscopic scale. Both of these factors are considered controls of functions.

No standardized terms have been defined to describe all that happens at the different geographic, temporal, or spatial scales. As a result, the authors of this synthesis have combined the terms and information used by several different authors to arrive at the list of factors in Table 2-2.

Table 2-2. Environmental factors that have been identified as controls of functions in wetlands. Most of the controls can occur at both the landscape scale and the site scale.

Environmental Factors that Control Functions in Wetlands	Scale at which Control Occurs
Physical structure of wetlands (soils, vegetation, rocks, etc.)	Site
Input of water (amount of water; maximum and minimum water levels)	Landscape and site
Fluctuations of water levels (frequency, amplitude, direction of flows)	Landscape and site
Input of sediment	Landscape and site
Input of nutrients	Landscape and site
Input of toxic contaminants	Landscape and site
Temperature	Landscape and site
Level of acid (pH)	Landscape and site
Concentration of salts	Mostly site
Connectivity to other habitats or ecosystems	Landscape
This table is a synthesis of the information presented by Winter (1983, 1986), LaBaugh et al. (1987), Winter and Woo (1990), Naiman et al. (1992), Brinson (1993), Brinson et al. (1995), Bedford (1999), Beechee and Bolton (1999), Gersib (2001), Adamus et al. (2001), Stanley and Grigsby (2003).	

The relationship between these environmental factors that control wetland functions and how they interact with human disturbances is shown conceptually in Figure 2-3.

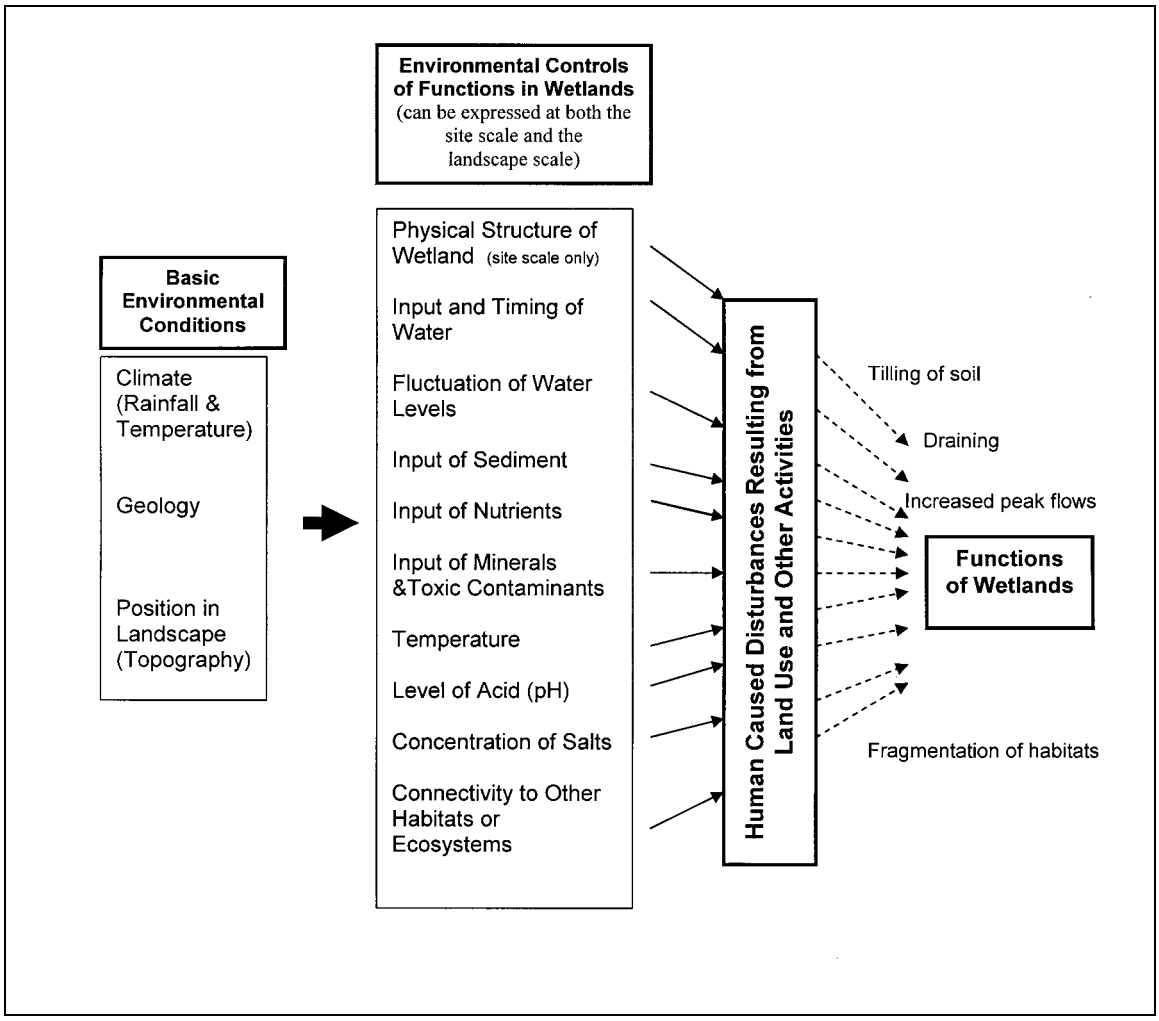

Figure 2-3. Diagram summarizing the environmental factors that control functions of wetlands and how they interact with human disturbances. The basic environmental conditions establish and determine the factors that control the functions of wetlands. The controls can occur at both the landscape and site scales. Human activities cause disturbances that affect these controls in many different ways and thereby alter the performance of wetland functions. The figure gives some examples of the disturbances.

2.2.4 Summary of Key Points

- The things that wetlands "do" are called wetland functions. There are many ways to define functions depending on specific needs for managing wetlands.
- Functions fall into three broad categories: hydrologic, biogeochemical, and maintenance of habitat and food webs.
- Society does not necessarily attach value, or equal value, to all functions.
- The functions that wetlands perform are controlled by environmental factors that occur in the broader landscape as well as within the wetland. The major controls

of function are climate; geomorphology and soils; the source of water; the movement of water, nutrients, other chemicals, and sediments; and energy in the form of sunlight.

- The factors that control wetland functions interact with each other and there are many feedback loops.
- In order to gain a basic understanding of the ecological significance of functions provided by wetlands, they must be evaluated within the context of the landscape in which they exist.

2.3 Classification of Wetlands in Washington as a Key to Understanding Their Functions

This section presents a brief discussion of systems that scientists have developed to group or classify wetlands nationally and in Washington State in order to better assess how they function. It begins with an overview of two classification systems—the Cowardin classification, commonly used to inventory wetlands across the country, and the hydrogeomorphic or HGM classification, which is used to characterize how wetlands function. Understanding how wetlands are grouped and classified is key to fully understanding how different types of wetlands in different areas provide different functions.

2.3.1 Commonly Used Classification Systems in Washington

The first commonly used classification system for wetlands was developed in 1979 by the U.S. Fish and Wildlife Service (Cowardin et al. 1979). The Cowardin system is hierarchical and includes several layers of detail for wetland classification that are based on:

- Water flow
- Substrate types
- Vegetation types
- Dominant plant species

The Cowardin classification system was developed to aid a national inventory of wetlands using aerial photographs (the U.S. Fish and Wildlife Service National Wetland Inventory or NWI). The wetlands in the state that can be identified from aerial photographs have been mapped using this classification system. The maps are available from the U.S. Fish and Wildlife Service in a digital form for GIS (http://www.nwi.fws.gov/). This information is a useful starting point for developing inventories of wetlands at the local level and looking at wetlands at the scale of watersheds and river basins.

Methods for organizing our knowledge about wetlands have been called classifications, categorizations, characterizations, ratings, assessments, and evaluations. These groupings are meant to indicate the type of information a method provides. Unfortunately, the scientific community has been inconsistent in the use of these terms. Users of methods developed for analyzing wetlands should be aware of some of these problems with definitions. See Appendix E for further discussion.

Although the Cowardin classification is useful in developing wetland inventories from aerial photographs and incorporates some landscape factors, it was not designed to help understand how functions differ among wetlands. A more recent system of classification, called the hydrogeomorphic (HGM) classification (Brinson 1993a), was developed to specifically address differences in how various wetlands function. This classification method was chosen by the statewide wetland technical committee that guided the development of the Washington wetland function assessment methods (Hruby et al. 1999).

The HGM classification is based on (Brinson 1993a):

- The position of the wetland in the landscape (geomorphic setting)
- The water source for the wetland
- The flow and fluctuation of the water once in the wetland

These are three of the major environmental factors that control wetland functions as described in Section 2.2. Classifying wetlands based on how they function narrows the focus of attention to a specific type of wetland, the functions that wetlands within the type are most likely to perform, and the landscape and ecosystem factors that are most likely to control how wetlands in that type function.

The HGM classification also uses the concept of grouping wetlands by geographic units (domains and regions) in which some of the controls of functions that occur at the landscape scale are similar. The assumption is that many of the functions performed by wetlands are also similar. Section 2.3.2 describes these "landscape" units in relation to the HGM classification system.

The highest category in the HGM classification (called "class") is defined nationally (Table 2-3) and is based on the geomorphic setting of the wetland (Brinson 1993a, Smith et al. 1995). Not all geographic units (domains and regions) contain all the wetland classes possible.

Within a region, wetland classes can be further divided by local experts into wetland subclasses and sub-subclasses (sometimes called "families" of wetlands) based on other geomorphic or hydrologic characteristics. The wetland experts in each region can, therefore, tailor the classification to address differences in the performance of functions by different wetland types in their region (Smith et al. 1995).

Geographical areas to which this classification system is applied in Washington and a description of the HGM classes in the state are described in Section 2.3.4.

Table 2-3. Characteristics of wetland classes in the hydrogeomorphic classification (from Brinson 1993a).

Hydrogeomorphic Class	Dominant Source of Water	Dominant Hydrodynamics (Movement of Water)
Riverine	Overbank flow from channel, or hyporheic (underground) flow in floodplain	One direction, horizontal
Depressional	Surface runoff, or the "daylighting" of groundwater	Vertical
Slope	"Daylighting" of groundwater on slopes	One direction, horizontal
Lacustrine (Lake) Fringe	Lake water	Two directions, horizontal
Flats	Precipitation	Vertical
Tidal Fringe	Overbank flow from estuary	Two directions, horizontal

2.3.2 Geographical Differences in Wetland Functions

Because hydrogeologic settings and the controls of functions vary across the landscape, it is important to identify the geographic areas in which these factors are similar. This allows the grouping of wetlands that function similarly.

For example, two conferences on wetland functions in the mid-1980s highlighted some of the differences between wetlands on the West Coast and those in the rest of the country (Horner 1986). Specifically, wetlands on the West Coast are different for the following reasons (Horner 1986 citing Zedler 1985):

- Drainage areas to West Coast wetlands are often smaller than those on the East Coast.
- The coastal plain, with some exceptions, is not as large on the West Coast.
- Soils in the West Coast region are often high in clay.
- Conditions in a watershed are often highly erosive on the West Coast because of the steep topography.
- Precipitation varies more seasonally on the West Coast than east of the Rocky Mountains.

Even within Washington, the diverse areas of the state support many kinds of wetlands that vary in functions. For example, vernal pools on the scablands differ greatly from the floodplain marshes along the Snoqualmie River, and wetlands that formed in the potholes created by glaciers have different functions from those found along the shores of salt lakes in the Grand Coulee (Hruby et al. 2000).

Through the Washington State Wetland Function Assessment Project, there has been a major effort over the last eight years to build on previous work and develop methods for assessing how wetlands function in different regions of the state. The methods are based on a formal process of quantifying the collective judgment of a group of local experts. This approach provides a scientific basis for rapid methods in the absence of rigorous, site-specific scientific studies (Hruby 1999).

A statewide technical committee was formed in 1994 to guide the technical components of the function assessment project. In addition, several assessment teams, composed of experts in different disciplines, developed methods for specific wetland types and areas of the state (Hruby et al. 1999, 2000). At present, methods for four wetland types in the lowlands of western Washington and three types in the Columbia Basin of eastern Washington have been completed. These documents are available on the project's web site (http://www.ecy.wa.gov/programs/sea/wfap/index.html).

Another major effort is currently underway to incorporate differences among geographic units and wetland functions into the Washington State Wetland Rating Systems for eastern and western Washington. The Washington State Department of Ecology is coordinating this effort, and teams of regional wetland experts and local government staff are providing technical expertise for developing drafts. A draft for eastern Washington is out for review. Work on revisions to the western Washington rating system is ongoing.

The domains and regions as identified by these teams of regional experts are described in the next section.

2.3.3 Wetland Regions in Washington

Wetlands in Washington are grouped first into *domains* and *regions* based on climate and other landscape features, then into *classes* by geomorphic setting, and finally into *subclasses* and *families* by the sources of water for the wetland and how that water moves (Hruby et. al. 1999, 2000, Hruby 2003). These are some of the primary controls of wetland functions as described earlier. This section focuses on the wetland domains and regions. Section 2.3.5 describes the wetland classes and Section 2.3.6 the subclasses for Washington State.

The wetlands in Washington were divided into two ecological domains, East and West, when the Washington State Wetland Rating System was first developed (Ecology 1991). The teams of wetland experts who are currently revising the rating system have kept this division (Hruby 2003). At this highest level, the domains are based on the national classification of ecosystems (called "ecoregions") developed by federal agencies (Bailey 1995). Wetlands on the west side of the Cascade Crest fall within the domain called "Humid Temperate" and those on the east side are in the "Dry" domain.

The term *ecoregion* was coined by J.M. Crowley (1967) and popularized by Robert J. Bailey (1976) to define a classification of ecosystems in the United States. Ecoregions are generally considered to be regions where climatic conditions are similar. As a result, the ecosystems there, including wetlands, are relatively homogeneous (Omernik and Gallant 1986). The concept was developed to help resource managers better understand regional differences in the environmental factors that maintain ecosystems and the relative importance of different factors that can change ecosystems (Omernik and Gallant 1986). The local maps of the ecoregions and their definitions are continually being updated by the U.S. Environmental Protection Agency laboratory in Corvallis, Oregon. The latest maps of ecoregions are available on the web at http://www.epa.gov/bioindicators/html/ecoregions.html.

The experts developing the Washington State Wetland Function Assessment Methods further divided the domains into smaller regions because the domains are too coarse a division for understanding how wetlands function in the state in a more detailed way. At present there are five regions in the state (Figure 2-4) including three regions in the eastern domain and two in the western domain:

- Eastern domain:
  - Montane
  - Columbia Basin
  - Lowlands of Eastern Washington
- Western domain:
  - Montane
  - Lowlands of Western Washington

These regions of Washington are also linked to the national classification of ecoregions developed by several federal agencies. The boundaries of the "regions" used in Washington, however, in some cases include parts of multiple "ecoregions" defined at the national level. For example, the region defined as the Lowlands of Western Washington includes areas of four ecoregions, whereas the region defined as the Columbia Basin is the same as the Columbia Plateau Ecoregion defined at the national level. The geographic extent of the Lowlands of Western Washington includes portions of three ecoregions within the "Humid Temperate" domain: the Coast Range, the Puget Lowlands, and the Willamette Valley (Hruby et al. 1999). Characteristics of these ecoregions are detailed in Omernik and Gallant (1986). The geographic extent of the Columbia Basin region, however, is the same as the "Columbia Basin Ecoregion" identified by Omernik and Gallant (1986).

At present, final definitions have been developed only for the Lowlands of Western Washington and the Columbia Basin regions because these are the only two regions for which methods to assess wetland functions have been developed. The Montane regions (east and west of the Cascades) and the Lowlands of Eastern Washington have been defined with less detail until assessments have been developed. Generally the Montane regions include areas above 3,000 feet (915 m) elevation, and the Lowlands of Eastern Washington includes all other areas in the "Dry" domain, outside the Columbia Basin, and below 3,000 feet (915 m) elevation.

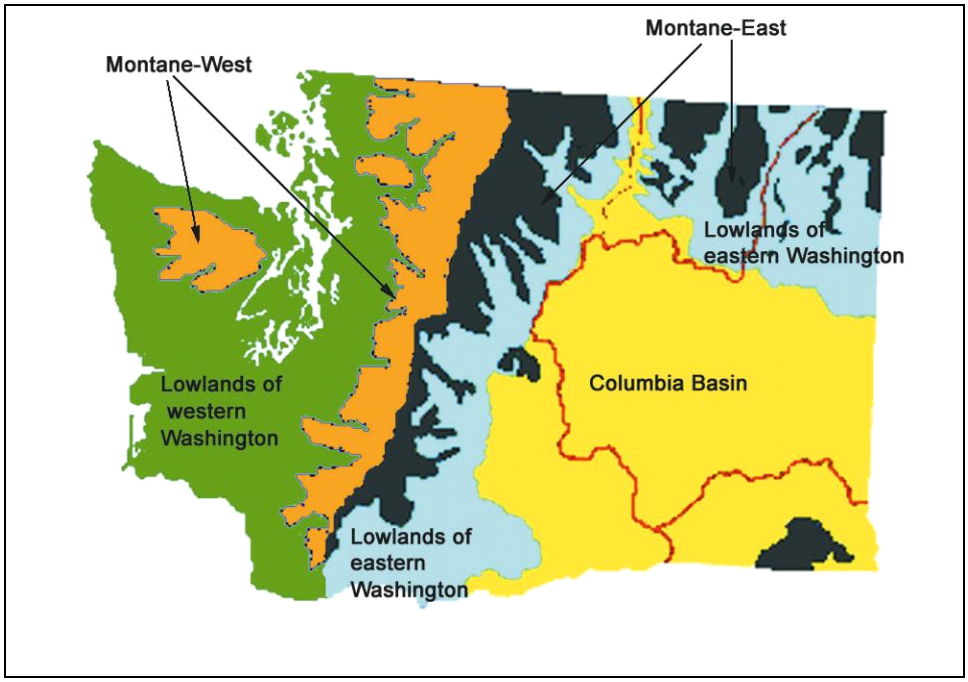

Figure 2-4. Regions in Washington used for classifying wetlands.

2.3.4 Description of the Wetland Classes for Washington

A brief description of wetlands in the different classes in Washington is given below. More detailed descriptions are available in Hruby et al. (1999, 2000).

2.3.4.1 Riverine Wetlands

The distinguishing characteristic of riverine wetlands in Washington is that they are frequently flooded by overbank flow from a stream or river (Hruby et al. 1999). Riverine wetlands are found in a valley or adjacent to a stream channel (Figure 2-5). They lie in the active floodplain of a river or stream and have important links to the water dynamics of the river or stream. The flooding waters are a major environmental factor that structures the ecosystem in these wetlands and controls wetland functions.

Figure 2-5. Riverine wetlands are located in active floodplains where overbank flooding of the river or stream structures the wetland ecosystem and controls its functions.

2.3.4.2 Depressional Wetlands

Depressional wetlands occur in topographic depressions that have closed contours on three sides (Figure 2-6). Elevations within the wetland are lower than in the surrounding landscape. The shapes of depressional wetlands vary, but in all cases the movement of surface water and shallow subsurface water is toward the lowest point in the depression. The depression may have an outlet, but the lowest point in the wetland is somewhere within the boundary, not at the outlet. (Hruby et al. 1999.)

Figure 2-6. Depressional wetlands are located in topographic low areas that are closed on at least three sides. They may or may not have an outlet.

2.3.4.3 Slope Wetlands

Slope wetlands (Figure 2-7) occur on hill or valley slopes where groundwater "daylights" and begins running along or immediately below the soil surface. Water in these wetlands flows only in one direction (down the slope) and the gradient is steep enough that the water is not impounded. The "downhill" side of the wetland is always the point of lowest elevation in the wetland. (Hruby et al. 2000.)

Figure 2-7. Slope wetlands are located on slopes where groundwater daylights and runs at or just below the soil surface.

2.3.4.4 Lacustrine Fringe (Lake-Fringe) Wetlands

Lacustrine fringe wetlands in Washington are found along the edges of deeper bodies of water such as lakes or reservoirs (Figure 2-8). These wetlands occur at the margin of topographic depressions in which surface water covers more than 20 acres (8 ha) and is deeper than 7 feet (2 m) in western Washington or 10 feet (3 m) in eastern Washington. The dominant surface water movement in lacustrine fringe wetlands has a horizontal component due to winds or currents, but there may also be a corresponding vertical component resulting from wind or seasonal water fluctuations. (Hruby et al. 1999, 2000.)

Figure 2-8. Lacustrine fringe wetlands are found along the edge of large bodies of water, such as lakes.

2.3.4.5 Flats Wetlands

Flats wetlands are rare in Washington. They occur in topographically flat areas that are hydrologically isolated from surrounding groundwater or surface water. The main source of water in these wetlands is precipitation. They receive virtually no groundwater discharge or surface runoff from areas outside the wetland boundary. This characteristic distinguishes them from depressional and slope wetlands. (Hruby et al. 1999.)

2.3.4.6 Tidal Fringe Wetlands

Tidal fringe wetlands occur along the coasts and in river mouths to the extent of tidal influence. The dominant source of water is from the ocean or a river that empties into the ocean; therefore these wetlands can be fresh or saline. The unifying characteristic of this class is the hydrodynamics. All tidal fringe wetlands have water flows dominated by tidal influences and water depths controlled by tidal cycles. (Hruby et al. 1999.) This document does not address tidal fringe wetlands, as stated in Chapter 1.

2.3.5 Subclasses of Wetlands in Washington

Developing the HGM classification for Washington is an ongoing process, and not all subclasses for wetlands in the different regions have been defined. The wetland subclasses and families that have been defined in the four regions of Washington (as of February 2003) are shown in Table 2-4.

Although the HGM classification for wetlands in the state is not yet complete, the categories listed in Table 2-4 provide a useful tool to help separate wetlands into different types.

Table 2-4. Subclasses and families of wetlands in different regions of Washington State (Hruby et al. 1999, 2000).

Class	Subclasses and Families by Region			
	Lowlands of Western WA	**Lowlands of Eastern WA**	**Columbia Basin**	**Montane (East and West)**
Riverine	Impounding Flow-through	ND	ND	ND
Depressional	Outflow Closed	ND	Alkali Freshwater Long-duration Short-duration	ND
Slope	ND	ND	ND	ND
Flats	ND	Probably does not occur in the region.	Probably does not occur in the region.	ND
Lacustrine (lake) Fringe	ND	ND	ND	ND
Tidal Fringe	Salt Water Fresh Water	Does not occur in the region.	Does not occur in the region.	Does not occur in the region.
ND = Subclasses in the region have not yet been defined.				

2.3.6 Summary of Key Points

- The physical structure and functions of wetlands vary by region. The diverse regions of Washington support many kinds of wetlands that provide different functions. These differences are being documented in the wetland functions assessment methods and rating systems for Washington State.
- Wetlands in Washington are grouped first into domains and regions based on climate, then by geomorphic setting, and finally by the sources of water for the wetland and how that water moves.
- Hydrogeomorphic classes in Washington State include riverine, depressional, slope, lacustrine (lake) fringe, flats, and tidal fringe. Subclasses and families of wetlands are also defined by region.

2.4 Overview of Wetland Functions in Washington State

As described in Section 2.2.4, our current knowledge about wetland functions in different regions of Washington and among different HGM classes is based largely on the work of experts involved in developing the function assessments and ratings for wetlands in the state (Hruby et al. 1999, 2000, Hruby 2003). Experts have developed methods to assess functions of riverine and depressional wetlands in several regions of the state. They have not discussed or identified the functions of freshwater wetlands in the flats, slope, tidal fringe, or lacustrine fringe classes, nor any functions of wetlands in the Montane regions.

As mentioned in Section 2.2.2 there are many ways to group wetland functions. Functions that are currently defined for the state are listed below. The definitions are compiled from Hruby et al. (1999, 2000) and Hruby (2003). Not all wetlands in a region, class, or subclass perform all of these functions. A more detailed description of each function is given in Section 2.5. As noted previously, functions are coarsely grouped into three main categories, those that improve water quality, those related to water regime in a watershed, and those that pertain to wildlife habitat.

> The functions selected for the Washington State Wetland Function Assessment Methods and the rating system are narrowly defined to provide a level of specificity that is important to decision-makers. The list of functions defined here does not represent all the functions performed by wetlands in the state. It does, however, represent the functions that were determined to be valuable and that need to be considered when managing wetlands (Hruby et al. 1999, 2000, Hruby 2003).

2.4.1 Functions Related to Improving Water Quality

Removing Sediment: This function is defined in terms of the processes and characteristics that retain sediment within a wetland and prevent its downstream movement. A wetland performs this function if there is a net annual decrease of sediment load to downstream surface waters.

Removing Nutrients/Phosphorus: This function is defined in terms of the processes and characteristics within a wetland that remove phosphorus present in surface waters and prevent its movement into surface waters and groundwater.

Removing Nutrients/Nitrogen: This function is defined in terms of the processes and characteristics within a wetland that remove dissolved nitrogen present in surface waters or groundwater and prevent its further movement into surface waters or groundwater.

Removing Metals and Toxic Organic Compounds: This function is defined in terms of the processes and characteristics within a wetland that retain toxic metals and toxic

organic compounds coming into the wetland and prevent their movement into surface waters and groundwater.

2.4.2 Functions Related to Maintaining the Water Regime in a Watershed (Hydrologic Functions)

Reducing Peak Flows: This function is defined in terms of the processes and characteristics within a wetland by which the peak flow in a watershed can be reduced during major storm or snowmelt, events that would otherwise cause flooding.

Decreasing Erosion: This function is defined in terms of the processes and characteristics within a wetland that detain high flows during storms and reduce the duration of erosive flows, thus decreasing downstream erosion in streams. This definition was developed for riverine and depressional wetlands. Wetlands along the shores of lakes (Jude and Pappas 1992) also protect resources from erosion but in a different way. For wetlands classed as lacustrine fringe, the function can be called Dissipation of Erosive Forces. This is defined as the processes by which wetlands reduce wave and current energies, thus decreasing erosion of shorelines.

Recharging Groundwater: This function is defined in terms of the processes and characteristics within a wetland that allow surface water to infiltrate into the groundwater system.

2.4.3 Functions Related to Habitat

General Habitat: This function is defined in terms of the processes and characteristics within a wetland that indicate a general suitability and opportunity as habitat for a broad range of species. A suitable habitat for a suite of different fauna can be provided by a broad range of structures, vegetation, and interspersion of "habitat" types within the wetland and the upland habitats contiguous to a wetland. Characteristics in a wetland can be quite different and continue to provide highly suitable conditions for a range of species.

Habitat for Invertebrates: This function is defined in terms of the processes and characteristics within a wetland that help maintain a high number of invertebrate species.

Habitat for Amphibians: This function is defined in terms of the processes and characteristics within a wetland that contribute to the feeding, breeding, or refuge needs of amphibian species.

Habitat for Anadromous Fish: This function is defined in terms of the processes and characteristics within a wetland that contribute to the feeding, breeding, or refuge needs of anadromous fish species.

Habitat for Resident Fish: This function is defined in terms of the processes and characteristics within a wetland that contribute to the feeding, breeding, or refuge needs of resident native fish.

Habitat for Wetland-Associated Birds (called Aquatic Birds in the methods for eastern Washington): This function is defined in terms of the processes and characteristics within a wetland that provide habitats or life resources for species of wetland-associated birds. Wetland-associated bird species are those that depend on aspects of the wetland ecosystem for some part of their life needs: food, shelter, breeding, or resting.

Habitat for Wetland-Associated Mammals (called Aquatic Mammals in the methods for eastern Washington): This function is defined in terms of the processes and characteristics within a wetland that support one or more life requirements of aquatic or semi-aquatic mammals.

Richness of Native Plants: This function is defined in terms of the degree to which the wetland provides a habitat for many different native plant species.

Supporting Food Webs (also called Primary Production and Export in the methods for western Washington): This function is defined in terms of the processes and characteristics within a wetland that support complex food webs within the wetland and surrounding ecosystems through the export and assimilation of the primary productivity of the wetland. The function combines three major ecosystem processes: primary production, secondary production, and export of production.

2.4.4 Summary of Key Points

- Wetland functions are currently defined for Washington State in a relatively narrow manner to facilitate better wetland management and regulation by decision-makers.
- Wetland functions defined in Washington fall into three general groups: functions related to improving water quality, functions related to the water regime in a watershed (hydrologic functions), and functions related to habitat.
- Not all wetlands in a region, class, or subclass perform all functions.

2.5 How Wetlands Perform Functions in Washington State

Table 2-5 summarizes the information on the functions that are, or are not, performed by the different freshwater wetland classes in Washington State. The following sections synthesize information available about each function and how the different wetland types in the state perform that function.

Table 2-5. Functions performed by wetlands in different HGM classes in Washington.

Key to symbols used in table:
P = Functions are performed
N = Functions are not performed
NS = Functions are probably not performed to any significant degree

Functions	Riverine	Depressional	Slope	Lacustrine Fringe	Flats
Improving Water Quality					
Removing Nutrients	P	P	P	P	NS
Removing Sediment	P	P	P	P	NS
Removing Metals/Toxic Organic Compounds	P	P	P	P	NS
Hydrologic					
Reducing Peak Flows	P	P	P	N	NS
Decreasing Downstream Erosion/Dissipating Erosive Forces	P	P	P	P	NS
Recharging Groundwater	P	P	N	N	NS
Food Webs and Habitat					
General Habitat	P	P	P	P	P
Habitat for Invertebrates	P	P	P	P	P
Habitat for Amphibians	P	P	P	P	P
Habitat for Anadromous Fish	P	P	N	P	N
Habitat for Resident Fish	P	P	N	P	N
Habitat for Wetland-Associated Birds	P	P	NS	P	P
Habitat for Wetland-Associated Mammals	P	P	NS	P	P
Plant Richness	P	P	P	P	P
Support Food Webs	P	P	P	P	P

Data compiled from Hruby et al. (1999, 2000), Hruby (2003).

2.5.1 Functions that Improve Water Quality

Wetlands greatly influence the quality of water in a watershed. They help improve water quality, including that of drinking water, by intercepting surface runoff and removing or retaining inorganic nutrients, processing organic wastes, and reducing suspended sediments before they reach open water. For example, as runoff water passes through wetlands, the wetlands retain or process excess nitrogen and phosphorus, decompose organic pollutants, trap toxic metals, and trap suspended sediments that would otherwise clog waterways and affect the development of fish and amphibian eggs (Sipple 2002).

2.5.1.1 Removing Sediment

Sediment may enter wetlands in direct runoff from surrounding areas, as windblown dust, or in streams or rivers that flow through the wetland. Sediments deposited in wetlands are removed from surface flows, thereby improving water quality down-gradient. A wetland will perform this function only if surface water contaminated with sediment actually enters the wetland.

Some general properties may be applied to all wetlands with respect to their ability to remove sediments (Phipps 1986). Within a given wetland, the deposition of sediment depends on several factors including (Phipps 1986, Johnston 1991, Fennessy et al. 1994, Gilliam 1994):

- Residence time of the water
- Wind and wave action
- Size and amount of incoming sediment
- Vegetation

Generally, a high residence time for the water and filtration by vegetation are the major processes by which sediment is removed from surface water (Fennessy et al. 1994). Filtration is the physical adhesion and cohesion of sediment facilitated by vegetation (Adamus et al. 1991). The size of the particles that settle out is directly related to the increase in settling time achieved in the wetland (Adamus et al. 1991).

Typically wetland vegetation traps 80 to 90% of sediment from runoff entering the wetland (Johnston 1991, Gilliam 1994). Other studies have found that open water may be as effective, or more effective, than vegetation in trapping sediments (Fennessy et al. 1994).

Wetlands can be disproportionately important for sediment retention compared to other components of the landscape (Adamus et al. 1991). Less than 65% of the sediment eroded from uplands exits watersheds that contain wetlands (Johnston 1991). Another way to consider the importance of wetlands for removing sediments in a watershed is to analyze how much wetland area is needed to effectively remove sediments. Fennessy et al. (1994) report the following:

- Watersheds in Wisconsin with only 5% of their area in wetlands trapped up to 70% of the sediment in the system (Novitzki 1979).
- In a North Carolina watershed, more than 20% of the total sediment deposition occurred in wetlands that represented only 11% of the area (Adamus 1988).

The importance of any wetland for improving water quality depends, however, on the amount of sediment pollution in the watershed. Watersheds in which human activities loosen the topsoil (agriculture, development, and logging) are prone to have high

sediment loadings. Wetlands in these watersheds are very important for maintaining water quality (National Research Council 1995).

Removal of Sediment by Wetlands of Various Classes and in Different Domains and Regions

The way wetlands remove sediment is not judged to be different in the two major domains of the state (the east side and the west side of the Cascades) (Hruby et al. 1999, 2000). However, the processes by which wetlands in Washington remove sediments differ somewhat among the different wetland classes as described below.

Wetlands in the Flats Class

Wetlands in the flats class, in general, do not remove sediment because by definition their major source of water is precipitation that falls within the wetland itself (Brinson 1993a). There is no opportunity for sediment-laden water to enter the wetland. All other types of wetlands perform this function to some degree because they receive surface water from outside their boundaries, and the surface water is never completely free of sediments.

Wetlands in the Depressional Class

Depressional wetlands that hold back all the surface water coming in (that is, those without a surface outlet) trap all the sediment they receive. Such wetlands are very effective at this aspect of water quality improvement wherever they are found in Washington (Hruby et al. 1999, 2000).

The removal of sediment in depressional wetlands with an outflow depends on how effectively they slow the water and allow settling, as well as the density of the vegetation that filters the incoming water. The same processes are present in depressional wetlands of both eastern and western Washington (Hruby et al. 1999, 2000).

Wetlands in the Lacustrine Fringe Class

Wetlands along the shores of lakes (lacustrine fringe) trap and retain suspended sediment by anchoring the shoreline, reducing resuspension of bottom mud by wind mixing, and slowing water velocities (Adamus et al. 1991). Even aquatic bed vegetation, which typically provides less resistance to water flow than emergent or woody plants, may reduce water movement enough to induce settling (Adamus et al. 1991).

Wetlands of this class have not yet been subjected to the thorough analysis required for developing a function assessment method. More definitive conclusions about Washington wetlands are, therefore, not available. However, no evidence has been reported that would negate the observations made in lacustrine wetlands in other parts of the U.S. that were reviewed by Adamus et al. (1991).

Wetlands in the Slope Class

Slope wetlands by definition (Brinson 1993) do not impound surface water. Removing sediment by increasing the residence time of water, therefore, is not a significant factor in this class of wetlands. Unpublished data collected during the calibration of the eastern Washington wetland rating system (Hruby 2003) suggest that slope wetlands may still play a role in removing sediment.

Slope wetlands in eastern Washington have vegetation that is usually thicker than the vegetation in the surrounding uplands (Figure 2-9). This vegetation acts like a filter to trap sediments coming from further upslope because it provides more resistance to the water flowing down the hillside (Hruby 2003).

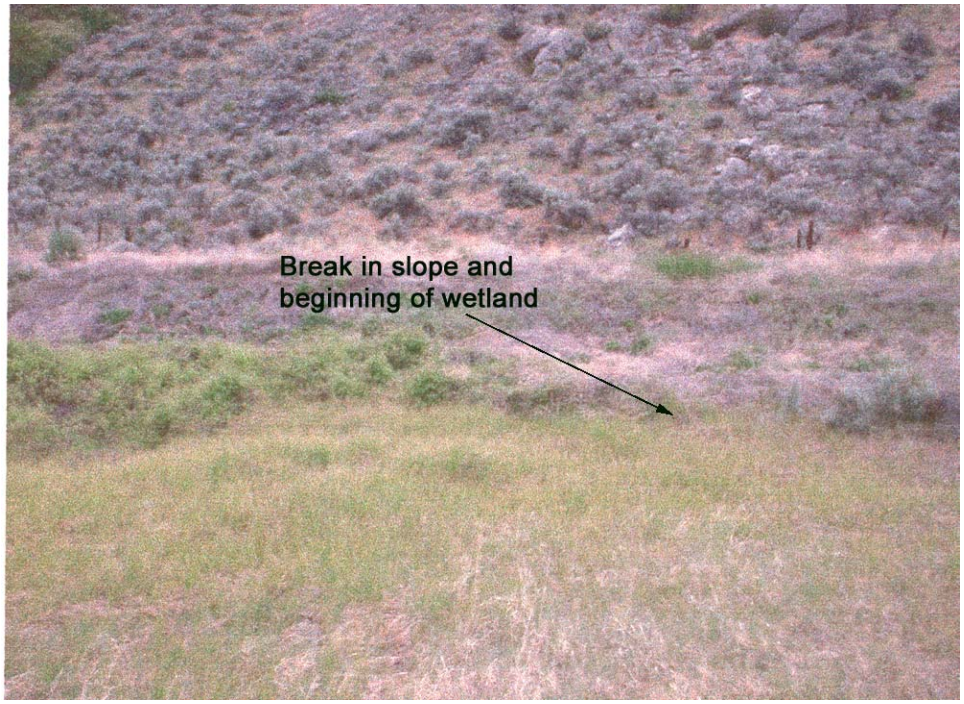

Figure 2-9. Slope wetland in the Columbia Basin that formed at a break in the slope. It has dense emergent plants that can trap sediment coming from the upslope areas.

Slope wetlands in western Washington have not yet been analyzed in terms of their potential to remove sediments, and it is not possible to report if similar processes and structure are found there. Models for assessing slope wetlands have, however, been developed for the Willamette Valley in Oregon. Two characteristics of slope wetlands identified there that contributed to the retention of sediments were the amount of ground covered by vegetation and the relative area of the wetland covered in hummocks (Adamus and Field 2001).

Wetlands in the Riverine Class

The removal of sediment in riverine wetlands is a somewhat different process. The vegetation and depressions within these wetlands trap sediment, but sediments are eroded

by floods that recur every few years. The function of riverine wetlands is to stabilize sediment during the period between floods (Adamus et al. 1991).

Phipps (1986) stated that the efficiency of sediment trapping by riverine wetlands in the Pacific Northwest has not been measured. This conclusion is still valid today, since no studies were found that quantified this function. The process of trapping sediments is still judged to be an important function on a watershed scale in Washington State (Hruby et al. 1999) and was modeled during the development of function assessment methods. The characteristics of riverine wetlands that were judged important in removing sediments were as follows (Hruby et al. 1999):

- How much the stream or river meanders through the wetland
- How wide the wetland is relative to the width of the stream
- How much of the wetland is covered in vegetation that can act as a filter
- The amount of constriction in the outlet (if the wetland has an outlet)

2.5.1.2 Removing Phosphorus

Phosphorus can enter wetlands with suspended solids or as dissolved phosphorus. It is usually transported attached to particles rather than dissolved in the water (Raisin and Mitchell 1995). The major processes by which wetlands keep phosphorus from going farther downstream are (Mitsch and Gosselink 2000):

- The trapping of sediment on which phosphorus is adsorbed
- The removal of dissolved phosphorus by adsorption to soils that are high in clay content or organic matter

Wetlands that are effective at trapping sediments, therefore, are also effective at removing phosphorus. The discussion in Section 2.5.1.1 on the classes of wetlands that are effective at removing sediments also applies to removing phosphorus (Hruby et al. 1999).

The adsorption of phosphorus on soils is not permanent. Certain conditions during periods of extensive anoxia (lack of oxygen) may release phosphorus into the overlying waters (Reddy and Gale 1994, Adamus et al. 1991). In general, however, wetlands are a sink for phosphorus in watersheds (Adamus et al. 1991).

Other data also show that phosphorus retention in wetlands is highly variable. Whigham et al. (1988) concluded that wetlands where waters had extensive contact with vegetation and/or organic litter were the most effective at phosphorous removal. Forested wetlands were only effective during flood events (when there was contact between waters and vegetation and more sediment deposition occurred). They found open water, lacustrine systems to be the least effective at phosphorous removal. Johnston et al. (1997) observed that a wetland may remove phosphorus from incoming waters during one part of the year

but at other times of year it may add phosphorus to water leaving the wetland. They hypothesized that the release of phosphorus from a wetland is due to the leaching of phosphorus from dying wetland vegetation.

Phosphorus Removal by Wetlands of Various Classes and in Different Domains and Regions

The way wetlands remove phosphorus is similar in wetlands in the two domains of the state (the east and west sides of the Cascades). Wetlands that are effective at trapping sediments are also effective at removing phosphorus (Hruby et al. 1999, 2000). This conclusion is based on data showing that most of the phosphorus entering a wetland is bound to sediment (Dortch 1996, Mitsch et al. 1995, Mitsch and Gosselink 2000). Refer to the previous discussion of sediment removal.

Phosphorus entering a wetland in a dissolved form can also be retained because it binds to clay and organic soils. The HGM classification does not separate wetland types by soil content (Brinson 1993a), so the presence of clay or organic soils is not specific to a particular wetland class. As a result it is not possible to differentiate this function between wetland types. In the absence of research to the contrary, it can be assumed that wetlands in all domains and regions of the state and in all wetland classes have the potential to remove phosphorus if they contain organic or clay soils.

Mechanisms of phosphorus removal

The following discussion from North Carolina State University summarizes the scientific literature on the ways in which wetlands remove and process phosphorus. (North Carolina State University n.d.).

Phosphorus removal from water in wetlands occurs through adsorption by aluminum and iron oxides and hydroxides; precipitation of aluminum, iron, and calcium phosphates; and burial of phosphorus adsorbed to sediments or organic matter (Walbridge 1993, Johnston 1991, Richardson 1985). Wetland soils can, however, reach a state of phosphorus saturation, after which phosphorus may be released from the system (Richardson 1985). Phosphorus export from wetlands is seasonal, occurring in late summer, early fall and winter as organic matter decomposes and phosphorus is released into surface water.

Dissolved phosphorus is processed by wetland soil microorganisms, plants, and geochemical mechanisms (Walbridge 1993). Microbial removal of phosphorus from wetland soil or water is rapid and highly efficient; however, following cell death, the phosphorus is released again. Similarly, for plants, litter decomposition causes a release of phosphorus. Burial of litter in peat can, however, provide long term removal of phosphorus. Harvesting of plant biomass is needed to maximize biotic phosphorus removal from the wetland system.

The potential for long-term storage of phosphorus through adsorption to wetland soil is greater than the maximum rates of phosphorus accumulation possible in plant biomass (Walbridge 1993, Johnston 1991). In alkaline wetlands, such as found in the West, phosphorus precipitates with calcium as calcium phosphate (Novotony and Olem 1994, Walbridge 1993). However, the presence of aluminum is the significant predictor of dissolved phosphorus sorption and removal from water in most wetland systems (Reddy and Gale 1994, Walbridge 1993, Richardson 1985). The capacity for phosphorus adsorption by a wetland, however, can be saturated in a few years if it has low amounts of aluminum and iron or calcium (Richardson 1985).

Wetlands along rivers have a high capacity for phosphorus adsorption because as clay is deposited in the floodplain, aluminum (Al) and iron (Fe) in the clay accumulate as well (Gambrell 1994). Thus floodplains tend to be important sites for phosphorus removal from the water column, beyond that removed as sediments are deposited (Walbridge 1993).

2.5.1.3 Removing Nitrogen

Wetlands in general act as sinks for nitrogen under both nutrient-enriched and un-enriched conditions (Adamus et al. 1991, Jansson et al. 1994). Nitrogen enters a wetland in the form of ammonium from animal wastes in runoff, as nitrate/nitrite from fertilizers in runoff and groundwater, or from air pollution (Adamus et al. 1991).

The efficiency of nitrogen removal is greater with longer retention times of the water, earlier plant community stages, and lower loading rates (Dorge 1984 as reported in Adamus et al. 1991). Wetlands are far more efficient at removing nitrogen from up-basin loading than either rivers or streams (Saunders and Kalff 2001), even though soluble nitrogen may be flushed out of wetlands at times of high flow (Johnston et al. 1990).

The major biochemical processes by which wetlands remove nitrogen are nitrification and denitrification. These respectively occur in alternating conditions where oxygen is present (aerobic) and oxygen is absent (anaerobic) (Johnston et al. 1990, Mitsch and Gosselink 2000, Vought et al. 1995, Saunders and Kalff 2001). Denitrification transforms the majority of nitrogen entering wetlands into nitrogen gas, causing between 70 and 90% to be removed from the aquatic system (Reilly 1991, Gilliam 1994).

In aerobic substrates, the bacteria *Nitrosomonas* can oxidize ammonium to nitrite. The bacteria *Nitrobacter* oxidizes nitrite to nitrate. This process is called nitrification. (Mitsch and Gosselink 2000.)

Nitrogen is completely removed from the aquatic system only by anaerobic bacteria that reduce nitrate to gaseous nitrogen during denitrification. The gaseous nitrogen volatilizes, and the nitrogen is eliminated as a water pollutant. Thus, the alternating reduced and oxidized conditions (anaerobic and aerobic respectively) of wetlands complete the nitrogen cycle and maximize denitrification rates (Johnston 1991).

Plants or microorganisms can use nitrate and ammonium for growth. Plant growth, however, does not really remove the nitrogen from the aquatic system because it becomes available again with the death of the plants or microorganisms that absorbed the nutrients (Adamus et al. 1991).

Nitrogen Removal by Wetlands of Various Classes and in Different Domains and Regions

The way wetlands remove nitrogen is similar east and west of the Cascades (Hruby et al. 1999, 2000, Hruby 2003). Furthermore, the HGM classification does not separate wetland classes by the amount of oxygen in the soils (Brinson 1993a). The presence of alternating cycles of anaerobic and aerobic conditions is not specific to wetland type or region. Therefore, it is not possible to differentiate this function between wetland types and regions.

In the absence of research to the contrary, it has been assumed that wetlands in all regions of the state and in all wetland classes may have the characteristics necessary to remove nitrogen (Hruby et al. 1999, 2000, Hruby 2003). Whether they do or not depends on the conditions found within the wetland, not on the type of wetland or its position in the landscape. The conditions that promote removal of nitrogen in wetlands of the state are seasonal inundation or saturation (Hruby et al. 1999, 2000). This indicates the soils alternate between aerobic conditions (when dry) and anaerobic conditions (when wet).

2.5.1.4 Removing Metals and Toxic Organic Compounds

The major physical, biological, and chemical processes by which wetlands reduce the amount of toxic materials moving into down-gradient waters are through sedimentation, adsorption, precipitation, plant uptake, and degradation (Adamus et al. 1991).

- **Sedimentation** is a major process by which wetlands remove toxic compounds. Wetlands that are effective at removing sediments are also effective at trapping toxicants. (See the previous discussion in Section 2.5.1.1.) For example, most heavy metals in urban runoff are adsorbed to sediment particles and are buried in sediment deposits within wetland soils (Canning referenced in Newton 1989).

- **Adsorption** of the compounds to the wetland soil is promoted by soils high in clay or organic matter (Adamus et al. 1991, Mitsch and Gosselink 2000). For example, wetlands can remove toxic metals from surface and groundwater if they contain clays, peat, aluminum, iron, and/or calcium (Gambrell and Trace 1994). Metals entering wetlands will bind to the negatively ionized surface of clay particles, or precipitate as inorganic compounds (metal oxides, hydroxides, and carbonates, depending on pH), or form a complex with humic materials (Gambrell and Trace 1994).

- Chemical **precipitation** is promoted by wetland areas that are inundated and remain aerobic, as well as those with pH values below 5 (Mengel and Kirkby 1982).

- **Plant uptake** of toxic compounds is maximized when there is significant wetland coverage by emergent plants (Kulzer 1990).

Removal of Toxic Compounds by Wetlands of Various Classes and in Different Domains and Regions

In Washington, the experts who developed assessment methods judged that wetlands that remove sediments effectively are also effective at removing toxic compounds (Hruby et al. 1999, 2000). This conclusion is based on data showing that some of the toxic compounds entering a wetland are bound to sediment (Mitsch and Gosselink 2000). Refer to the earlier discussion in Section 2.5.1.1.

Wetlands on the east and west sides of the Cascades function similarly in removing toxic compounds (Hruby et al. 1999, 2000, Hruby 2003). There may be some differences based on wetland class because some of the characteristics (such as effectiveness at trapping sediment) that are important for removing toxic compounds are dependent on the wetland class. Other differences do not depend on wetland class.

The HGM classification does not separate wetland types by the soils present or by how well they trap sediments (Brinson 1993a). The presence of clays, organic soils, aluminum, iron, or calcium in the soils is not specific to any wetland type. The function of removing toxic compounds, therefore, is not performed exclusively by a particular wetland class or subclass. In the absence of research to the contrary, it can be assumed

that wetlands in all regions of the state and in all wetland classes have the potential to remove toxic metals and organic compounds based on the composition of their soils.

Depressional wetlands and others with certain soil types, however, may perform this function at a higher level than most others. In general, the performance of this function may be better in wetlands with the following characteristics.

Wetlands with Clay Soils

Three types of soils are called "clay" according to Bluemle (1999):

- Those that consist of very finely ground rock formed by glaciers
- Those that were deposited in lakes and the ocean
- Those derived from the weathering of rocks in place

As mentioned above, wetlands with clay soils can remove toxic compounds because of the chemical properties of this type of soil. The scientific literature on the chemical properties of clays in relation to the adsorption of metals and organic pesticides, however, is based on the clays derived from weathered rocks such as bentonite, montmorillonite, and kaolinite (Fushiwaki and Urano 2001). There is little information on the chemical properties of clays derived from glacial activity or aquatic sediments.

County soil surveys (e.g., Debose and Klugland 1983) indicate that glaciers have played an important role in forming clays in western and northeastern Washington. Lacustrine (lake) and marine clays are also common in Whatcom County (NRCS 1992). These clays may contain chemically reactive minerals but data to confirm this assumption are not available. It is not possible, therefore, to make any definitive conclusions about the potential for all wetlands with clay soils to remove toxic compounds.

Wetlands with Volcanic Ash

Wetlands in the Columbia Basin often have a very fine layer of volcanic ash either from the Mt. St. Helens eruption in 1980 or the earlier Mt. Mazama eruption (based on unpublished data collected during the calibration of the methods for assessing wetland functions, Hruby et al. 2000). Volcanic ash that is washed or deposited into wet areas is in time transformed into bentonite clays (Bluemle 1999). Thus, the ash found in wetlands of the Columbia Basin may be performing as clays to remove toxic compounds, but this hypothesis does not have any supporting data at present.

Wetlands with Organic Soils

Wetlands with organic soils such as peat bogs and fens in Washington State have the necessary soil conditions to react with and adsorb toxic compounds. The effectiveness of these wetlands may further be increased because the water in these systems is often below pH 5 (based on unpublished data collected for the calibration of the Washington State Wetland Function Assessment Methods).

Wetlands in the Depressional Class

A number of the characteristics that enhance the removal of toxic compounds are present more often in depressional wetlands, although all depressional wetlands do not have these characteristics. A higher number of depressional wetlands have slower moving water and finer sediments compared to riverine or slope wetlands (Brinson 1993a). Wetlands in which water moves slowly are better at removing toxics than those in which water moves rapidly. Slow moving water allows more time for chemical processes to occur before the water moves out of the wetland. This promotes the settling of fine sediments and the formation of organic soils (North Carolina State University 2002).

Depressional wetlands in the state more often have organic soils than wetlands in the other classes (based on unpublished data collected by Ecology during the calibration of the Washington State Wetland Function Assessment Methods and the Wetland Rating System). Depressional wetlands, therefore, usually have a higher potential to remove toxic compounds than wetlands in the other classes. Their overall effectiveness at this function, however, also depends on the presence of toxic compounds in the watershed.

2.5.2 Functions Related to Maintaining the Water Regime in a Watershed (Hydrologic Functions)

Wetlands play an important role in the water regime of watersheds, as summarized by Sipple (2002):

> *Because of their low topographic position relative to uplands (e.g., isolated depressions, floodplains), wetlands store and slowly release surface water, rain, snowmelt, groundwater and flood waters. Trees and other wetland vegetation also impede the movement of flood waters and distribute them more slowly over floodplains. This combined water storage and slowing action lowers flood heights and reduces erosion downstream and on adjacent lands. It also helps reduce floods and prevents water logging of agricultural lands. Wetlands within and downstream of urban areas are particularly valuable in this regard, counteracting the greatly increased rate and volume of surface-water runoff from pavement and buildings.*
>
> *Because of their position on the landscape, wetlands at the margins of lakes, rivers, bays, and the ocean help protect shorelines and stream banks against erosion. Wetland plants hold the soil in place with their roots, absorb the energy of waves, and break up the flow of stream or river currents. The ability of wetlands to control erosion is so valuable that some states (e.g., Florida) are restoring wetlands in coastal areas to buffer the storm surges from hurricanes and tropical storms by dissipating wave energy before it impacts roads, houses, and other man-made structures.*

The following sections describe the characteristics of wetlands that reduce peak flow, reduce erosion, and recharge groundwater in Washington.

2.5.2.1 Reducing Peak Flows

Surface water that may otherwise cause flooding is stored to a greater degree in wetlands than typically occurs in terrestrial environments (Adamus et al. 1991). As a result, peak flows in streams and rivers are directly related to the total area of wetlands in the watershed, or to the area of wetlands in the headwaters of the system (National Research Council 1995). Wetlands reduce peak flows in streams and rivers by slowing and storing water in overbank areas and by holding back runoff that would otherwise flow directly downstream and cause more severe flooding (Reinelt and Horner 1995).

The function of reducing peak flows as defined in Washington State also includes the process of "floodflow desynchronization" (Hruby et al. 1999). This is a process that occurs at a larger, landscape scale. Desynchronization occurs when floodwaters are stored in many wetlands within the watershed. The release of water from these wetlands is staggered and gradual, resulting in more persistent flows but much lower peak flows (Adamus et al. 1991).

The characteristics of a wetland that indicate a potential to reduce peak flows include (Mitsch and Gosselink 2000):

- The volume of water storage (depth of water stored multiplied by wetland area)
- Proximity of the wetland to flood waters
- Location of the wetland (along a river, lake, or stream)
- Amount of flooding that would occur without the presence of the wetlands
- Lack of other upstream storage areas such as ponds, lakes, and reservoirs

Reduction in Peak Flows by Wetlands of Various Classes and in Different Domains and Regions

The importance of wetlands in reducing peak flows and how they perform this function differ in eastern and western Washington. This is a result of differences in the patterns of precipitation and snowmelt between the two areas (Hruby et al. 1999, 2000). The processes by which wetlands in Washington reduce peak flows also vary among wetland classes.

Wetlands of Western Washington

In **depressional wetlands of western Washington**, the characteristics within a wetland that reduce peak flows are the short-term storage capabilities of the wetland and the relative amount of flow captured from the upgradient contributing basin (Hruby et al. 1999). Short-term storage is often called "live-storage" by hydrologists. It is the amount

of water stored above the level of the outlet (if the wetland has one). Water stored below the outlet is called "dead-storage" and was not considered to be important in reducing peak flows in western Washington (Hruby et al. 1999). The "dead-storage" is usually filled by the time a flood event occurs and thus is not available to capture storm flows. Since most flooding events occur later in the fall, winter, and early spring, reductions in peak flow will occur only when a depressional wetland has some live-storage as well (Adamus et al. 1991, Hruby et al. 1999).

The expert teams who developed assessment methods for the state determined that the same assumption applies to the storage within the interstices of the soil (spaces between soil particles). Wetland soils in western Washington are usually saturated by the time most flood events occur, and storage in the soils was not judged to be important in reducing peak flows (Hruby et al. 1999) although it has been suggested as an important characteristic in other parts of the nation (Adamus et al. 1991).

Depressional wetlands with no outlet store all surface waters coming into them and therefore have the highest potential to reduce peak flows (Hruby et al. 1999).

In **riverine wetlands of western Washington**, the major characteristic that reduces peak flows is the storage provided by overbank areas (Hruby et al. 1999). As floodwaters rise, the waters overtop the banks of the river and fill the adjacent areas, many of which are riverine wetlands. The presence of a wide surface with an elevation at or near that of the river bank is the most important factor in reducing peak flows. As the flood waters overtop the banks they are slowed down and the height of the flooding is reduced because the excess water is stored in these wetlands longer than the duration of the peak flows (Adamus et al. 1991, Hruby et al. 1999).

The **lacustrine fringe, flats, and slope classes of western Washington** have not been analyzed relative to reducing peak flows. The information available suggests wetlands in the flats and slope class do not play a major role in this function. Wetlands in the flats class by definition do not receive any runoff from surrounding areas (Brinson 1993a). Their effectiveness at reducing peak flows is to store only the precipitation that falls within their boundaries.

Wetlands in the slope class do not provide storage because by definition they do not impound any surface water (Brinson 1993a). Water flows to the lowest point on the slope and is then discharged. The one role slope wetlands may play is to reduce the velocity of surface runoff by way of the thick vegetation often growing there. (See Figure 2-9 for an illustration.) The importance of vegetation on slopes in reducing flows has been well documented in studies of logging, though not specifically for slope wetlands (Lewis et al. 2001). It may be appropriate to assume that vegetation in slope wetlands plays the same role as vegetation in forested areas in reducing peak flows (Hruby 2003).

Wetlands of Eastern Washington

In **depressional wetlands of eastern Washington**, the characteristics within the wetland that reduce peak flows are the total storage capacity of the wetland and the relative amount of flow it captures from the upgradient contributing basin (Hruby et al. 2000).

The events that cause flooding in eastern Washington are different than in the western part of the state. Summer thunderstorms can cause flooding at times when most depressional wetlands are dry. As a result, the entire storage capacity of the wetland is available rather than just the "live-storage" (Hruby et al. 2000). Depressional wetlands with no outlet store all surface waters coming into them and therefore have the greatest potential to reduce peak flows.

Riverine wetlands in eastern Washington function in a fashion similar to those on the west side (Hruby 2003). Although function assessment methods have not been developed, the field work undertaken in calibrating the revised wetland rating system suggests that the major characteristic that reduces peak flows is also the storage provided by overbank areas (Hruby 2003). See the previous discussion of riverine wetlands in western Washington for a more detailed description of storage by overbank areas.

Wetlands in **the lacustrine fringe and slope class** have not been analyzed in eastern Washington for their ability to reduce peak flows. The information collected during the calibration of the eastern Washington rating system, however, suggests wetlands in these two classes provide this function but not at the same levels as riverine or depressional wetlands (Hruby 2003). Wetlands along the shores of lakes and reservoirs in eastern Washington tend to be small relative to the area of the lake (based on unpublished data, Hruby 2003). They have some capacity to store water as the water levels in a lake rise, but the extra amount stored is often very small compared to the storage in the lake itself.

Furthermore, many lakes and reservoirs in this region have controlled and manipulated outlets. This means that the reduction in peak flows is directly controlled by humans and not by ecological processes. It is not possible, therefore, to assess how well these wetlands function to reduce peak flows based on their characteristics without an understanding of the protocols used to regulate the water levels in each reservoir.

By definition, wetlands in the slope class do not provide storage because any water flows to the lowest point and then is discharged (Brinson 1993). However, their frequently dense vegetation reduces the velocity of surface runoff (see Figure 2-9) and thus can reduce peak flows somewhat. A wetland with dense vegetation will intercept more runoff and be more capable of reducing runoff velocity (and thus peak flows) than a wetland with less dense vegetation (Richardson and McCarthy 1994).

The importance of vegetation on slopes in reducing flows has been well documented in studies of logging (Lewis et al. 2001) though not specifically for slope wetlands. In eastern Washington the assumption is that vegetation in slope wetlands plays the same role as vegetation in forested areas in reducing peak flows (Hruby 2003).

2.5.2.2 Decreasing Downstream Erosion

The major process by which wetlands reduce downstream erosion is by slowing the velocity of water flowing downstream (Reinelt and Horner 1995, Adamus et al. 1991). The reduction in velocity depends on (Adamus et al. 1991):

- Channel constrictions that slow the flow of water
- Frictional resistance of the bottom
- Frictional resistance of vegetation

Jadhav and Buchberger (1995) state that the drag induced by plant stems increases with water velocity. This means that the relative reduction in velocity caused by plants increases as the speed of the water increases.

Reduction of Erosion by Wetlands of Various Classes and in Different Domains and Regions

The ways by which wetlands decrease erosion are somewhat different east and west of the Cascades. This is a result of the differences in the patterns of precipitation and snowmelt between the two areas (Hruby et al. 1999, 2000). The processes by which wetlands in Washington reduce erosion can also differ among wetland classes, as described below.

Wetlands of Western Washington

In **depressional wetlands of western Washington**, several characteristics influence a wetland's function in reducing water velocities (Hruby et al. 1999):

- Short-term storage capabilities of the wetland
- Characteristics of its outlet
- Amount of woody vegetation present
- Relative amount of flow captured from the upgradient contributing basin

Depressional wetlands with no outlet store all surface waters flowing into them. They have the greatest potential, therefore, to decrease erosion because no water leaves the wetland that could cause erosion (Hruby et al. 1999).

In **riverine wetlands of western Washington**, the major characteristic that reduces erosion is the amount of woody vegetation present that can provide a barrier to water flows (Hruby et al. 1999). As flood waters overtop the river banks, they are slowed down. The width of the wetland relative to the channel indicates how well the wetland can reduce velocity; the wider the wetland, the more water can spread out, becoming shallower and slowing down (Hruby et al. 1999).

Function assessment methods have not been developed for the **lacustrine fringe, flats, and slope classes in western Washington** and there is little information available on how these types of wetlands may perform this function. Wetlands in the flats class, however, are not expected to play a major role in this function. By definition, they do not receive any runoff from surrounding areas and therefore do not intercept waters that can cause erosion (Brinson 1993a).

Wetlands in the slope class, however, may decrease erosion to some degree because they often have thick vegetation relative to the surrounding uplands that reduces the velocity of surface runoff. Jadhav and Buchberger (1995) state that under dynamic conditions (high flows such as those found on slopes during storms) velocity is reduced by the drag induced by plant stems. Wetland detention time is therefore increased with vegetation density.

It can also be hypothesized that wetlands along the shores of lakes in western Washington (lacustrine fringe) may reduce erosion along the shore because of the vegetation they support. This would both anchor the shoreline and dissipate erosive forces (Adamus et al. 1991). Wetlands that have extensive, persistent (especially woody) vegetation provide protection from waves and currents associated with large storms that would otherwise penetrate deep into the shoreline (Adamus et al. 1991).

Wetlands of Eastern Washington

In **depressional wetlands of eastern Washington**, the characteristics within the wetland that decrease erosion are the total storage capacity of the wetland and the relative amount of flow captured from the upgradient contributing basin (Hruby et al. 2000). The events that cause erosion in eastern Washington are different than in the western part of the state. Summer thunderstorms can cause highly erosive flows at times when most depressional wetlands are dry (Hruby et al. 2000). As a result, the entire storage capacity of the wetland is usually available to reduce water velocities rather than just the "live-storage." Depressional wetlands with no outlet store all surface waters coming into them and therefore have the most potential to decrease erosive flows.

Riverine wetlands in eastern Washington function in a similar fashion to those on the west side (Hruby 2003). Although experts have not developed function assessments, the field work undertaken in calibrating the revised wetland rating system suggests that woody vegetation within the wetland is key in reducing erosive flows by slowing velocities during floods.

Function assessment methods for the **lacustrine fringe and slope classes** have also not been developed in eastern Washington. There is therefore no clear understanding of how they function to decrease erosion. It can be hypothesized, however, that wetlands of both classes can function to reduce erosion to some degree. (See the discussion for slope and lacustrine fringe wetlands in western Washington above.)

2.5.2.3 Recharging Groundwater

The recharge of groundwater is the movement of surface water, usually downward, into the ground. In wetlands, the function is described in terms of the wetland structures and processes that allow surface water to infiltrate into the groundwater system. Adamus et al. (1991) and the expert teams developing the Washington State Wetland Function Assessment Methods (Hruby et al. 1999, 2000) concluded that the movement of water into the ground depends primarily on:

- The elevation of the wetland relative to the groundwater
- The mass and pressure of water (pressure head) in the wetland
- The physical characteristics and frictional resistance of the sediments and strata underlying the wetland (hydraulic conductivity)

If the surface of the water in a wetland is groundwater, or the primary source of water to the wetland is groundwater (e.g., a seep), the wetland cannot "recharge" that groundwater. By definition, recharge occurs only if water from surface runoff infiltrates into groundwater.

The little information available on the potential for wetlands to recharge groundwater is contradictory. Adamus et al. (1991) conclude, from an extensive review of the literature, that four site-specific conditions determine how well a wetland performs this function:

- Groundwater flow rates under the wetland (linked to hydraulic conductivity)
- The storage capacity of the wetland (linked to the "pressure head" of water)
- Water movement within the wetland (linked to elevation relative to groundwater and hydraulic head)
- Evapotranspiration (linked to "pressure head" of water in the wetland)

This conclusion about these site-specific conditions was more recently confirmed by Hunt et al. (1996).

Adamus et al. (1991) were unable to find any patterns among wetland types or regions of the country. They also concluded that "for recharge, adjacent undeveloped uplands are usually, but not always, more important than wetlands."

Groundwater Recharge by Wetlands of Various Classes and in Different Regions

The characteristics within a wetland that result in the recharge of groundwater are the same for wetlands in both the eastern and western parts of the state. The potential for recharge in a wetland occurs when wetlands hold back precipitation and surface flows to create ponded areas. This ponded water then infiltrates into the groundwater system because of the "head" or pressure created by the depth of water on the surface. If the

hydraulic head created by upslope groundwater is greater than the hydraulic head created by the ponded water, recharge will not occur (Adamus et al. 1991).

Groundwater recharge occurs only in a subset of **depressional wetlands** and some **riverine wetlands** that impound and hold surface water. Wetland types that do not impound surface water do not have the potential to recharge groundwater (Hruby et al. 1999, 2000, Hruby 2003).

A new perspective on baseflow support

One aspect of groundwater recharge that is often attributed to wetlands in Washington is called "baseflow support." Wetlands are assumed to augment base flows in streams during the drier seasons because of the water they store. The information available, however, indicates this assumption is not valid in most cases. In a review of the literature up to 1991, Adamus et al. (1991) concluded that discharge from wetlands during the dry season is very small if it exists at all. In fact, Adamus and his colleagues concluded that some wetlands may reduce baseflow because of evapotranspiration. This generalization, however, may not be valid in peat wetlands that lie on top of permeable mineral soils since Reeve et al. (2000) found some support of baseflow from such peat wetlands in Canada and Minnesota.

In Washington, the teams of experts that developed the methods for assessing functions and the rating systems concurred with the original generalizations (Hruby et al. 1999, 2000, Hruby 2003). Surface outflow from the wetland was not judged to be an important factor in maintaining low flows in streams in Washington State. The wetland may be in a location where groundwater is discharged, but the source of this groundwater is not within the wetland itself. Thus, the discharge is not a function of the wetland; rather it is a function of the entire groundwater system as reported by Adamus et al. (1991).

2.5.3 Functions Related to Habitat

This section focuses on three aspects of wetlands as habitat:

- Structures and processes found within wetlands that make them an important habitat feature of the landscape
- The number and types of vertebrate species using wetlands in the Pacific Northwest
- Important features of wetlands that meet the habitat requirements of some groups of wetland-dependent species in Washington that were modeled in the Washington State Wetland Function Assessment Methods

The discussion is not subdivided by wetland class or domain and region of the state because habitat requirements differ widely for various species. Furthermore, habitat

requirements for a single species may even differ between locations (Adamus et al. 1991). Therefore, this literature review does not attempt to identify all the life requirements of all wildlife species that use wetlands in Washington. The intent of this synthesis is to identify some of the basic structures and processes in wetlands that are important habitat features.

2.5.3.1 Characteristics that Make Wetlands Important as Habitat

Wetlands are among the most productive ecosystems in the world, comparable to rain forests and coral reefs (Mitsch and Gosselink 2000, Sipple 2002). As a result, wetlands support numerous species from all of the major groups of organisms—from microbes to mammals (Sipple 2002). The support they provide for these organisms includes sources of food, shelter, and refuge. All of these aspects are generalized by the term "habitat."

General reviews of wetlands as habitat (Adamus et al. 1991, Mitsch and Gosselink 2000) conclude that physical and chemical characteristics (factors that control the suitability of a wetland as habitat) determine what plants and animals inhabit various wetlands, including:

- Climate
- Topography (landscape shape)
- Geology
- Nutrients
- Hydrologic regime (quantity and movement of water)

In addition, some of the larger organisms such as beaver and muskrats manipulate wetlands to create habitat suitable for themselves and other organisms, such as fish, amphibians, waterfowl, insects, and other mammals (Mitsch and Gosselink 2000).

Animals use wetlands to varying degrees depending upon the species involved. Some live only in wetlands for their entire lives; others require wetland habitat for at least part of their life cycles; still others use wetlands much less frequently, generally for feeding (Sipple 2002). Thus, species using wetlands can be divided into those that are "wetland-dependent" and those that are "wetland users."

Adamus et al. (1991) defined *wetland-dependent species* as those that: "(a) normally use wetlands exclusively for food and cover throughout most of their U.S. range and spend most of their lifetime within wetlands, or (b) would be extirpated from a large region if all wetlands were to be filled." The latter case includes species that may use wetlands for only part of their life cycles such as amphibians and many insects. The larvae of amphibians and many insects are aquatic even though the adults migrate out of the wetlands. The species are still considered to be wetland dependent because they could not survive without the presence of wetlands.

Species that are *wetland users* are those that use wetlands for occasionally obtaining some life requirements such as sources of drinking water, winter cover (e.g., white-tailed deer and ring-necked pheasants), or dispersal centers within urban areas (e.g., opossum) (Adamus et al. 1991).

Other terms used in the literature to reflect the different types of organisms using wetlands include the term "wetland-associated." For example, Kauffman et al. (2001) use the terms "closely associated," "associated," and "present" when describing the relationship between species and wetlands. The Washington State Wetland Function Assessment Method uses "wetland associated" (Hruby 1999).

Unfortunately, these terms are not defined so it is not possible to match the terms used in Adamus et al. (1991) with those used in Kauffman et al. (2001) and those used in the function assessment methods in Washington State.

Adamus et al. 1991 also state the following about how species use wetlands:

> *The degree of dependence by any given species on wetlands often varies greatly depending on the abundance and distribution of wetlands and on suitable alternative habitats within the region. For example, urban wetlands and riparian wetlands in the arid Southwest support species that, in other parts of their ranges, are much less likely to inhabit wetlands.*

Four general ecological features contribute to species richness and abundance in a landscape (Knutson and Naef 1997):

- Structural complexity
- Connectivity with other ecosystems
- Abundant food source and available water
- Moist and moderate microclimate

Wetlands have all of these attributes, especially wetlands that are linked to riparian areas and floodplains. The following sections describe each of these features in more detail.

Structural Complexity

Structural complexity is a term used to represent the variety of environmental characteristics that increase the number of niches for wildlife (Knutson and Naef 1997). These characteristics can include biological features such as a high richness of plant species or physical features such as open water, rocks, and mudflats. The interspersion in wetlands between open water and vegetation, or between types of vegetation, is important because the edges created between these elements (see Figure 2-10) increase the number of niches present (Adamus et al. 1991). Wetlands also often contain different vegetation communities within their boundaries that add structure (and therefore niches). For example, a higher interspersion of plant types is likely to support a higher diversity of invertebrates (Dvorak and Best 1982, Lodge 1985).

Figure 2-10. Features of a wetland that increase structural complexity. This wetland has open water and plants of different heights and different types (woody, herbaceous, aquatic bed) as well as snags and woody debris.

Riparian wetland systems in the semi-arid West often provide the only structurally complex habitat in regions dominated by open land or land cleared for agriculture (Adamus et al. 1991). This has also been found to be true in the semi-arid areas of eastern Washington, especially in the areas where rainfall is less than 12 inches per year. (Hruby et al. 2000). Figure 2-11 shows a wetland with high structural complexity in a semi-arid terrestrial environment that does not have much complexity.

Figure 2-11. Depressional wetland in the Columbia Basin showing a structurally complex ecosystem in a terrestrial environment with low complexity. The average annual rainfall at this site is 8 inches per year.

Connectivity to Other Aquatic Ecosystems

Many wetlands are linked to other ecosystems by surface water. Riverine wetlands form part of riparian corridors, depressional wetlands may be part of a small stream system or may be linked by surface water, and lacustrine fringe wetlands are connected to adjacent lakes. These connections provide corridors for the movement or migration of many animals (Kauffman et al. 2001). See Chapters 3 and 4 for further discussion of habitat connectivity and corridors to both aquatic and terrestrial habitats.

Abundant Food Sources

The wet and moist microclimate of wetlands and their rich soils lead to the enhanced growth of plants. Wetlands are known for their high primary productivity (production of plant material) and the subsequent movement of this "food" to adjacent aquatic ecosystems (Mitsch and Gosselink 2000). "Wetlands can be thought of as biological supermarkets" (Sipple 2002). For example, the number of invertebrates in small seasonal wetlands can exceed 700,000 animals per square meter (Leeper and Taylor 1998). Many of these invertebrates serve as food for larger predatory amphibians, reptiles, fish, birds, and mammals (Sipple 2002).

Moist and Moderate Microclimate

The presence of water and thick vegetation in wetlands results in a microclimate that is generally more moist and that has milder temperature extremes than the surrounding areas. These conditions provide a habitat that is desirable to many species, particularly amphibians, ungulates, and other large mammals during hot, dry summers and severe winters (Knutsen and Naef 1997).

2.5.3.2 Use of Wetlands by Vertebrates in Washington

Wetlands in the state have been shown to be critical in maintaining regional biodiversity. Although wetlands represent only 2.1% of the area of the state (Dahl 1990), over two-thirds of all terrestrial vertebrate species in Washington can be considered "wetland-dependent" or "wetland users" (Knutson and Naef 1997, Kaufmann et al. 2001). Kauffmann et al. (2001) have compiled information on vertebrate usage in two types of wetlands in Oregon and Washington they call "riparian" and "herbaceous." Their information, summarized below, indicates how important such wetlands are in regional biodiversity.

Reptiles and Amphibians

There are 59 species of reptiles and amphibians in Washington and Oregon. Two species of reptiles, the western pond turtle (*Clemmys marmorata marmorata*) and the painted turtle (*Chrysemys picta*), are wetland-dependent. Many more species of reptiles are wetland users. On the other hand, all but one species of amphibians are wetland-dependent and require an aquatic habitat for part of their life cycle (Kauffman et al. 2001). Figure 2-12 shows how many of the 59 species of reptiles and amphibians in the two states are found in riparian and herbaceous wetlands.

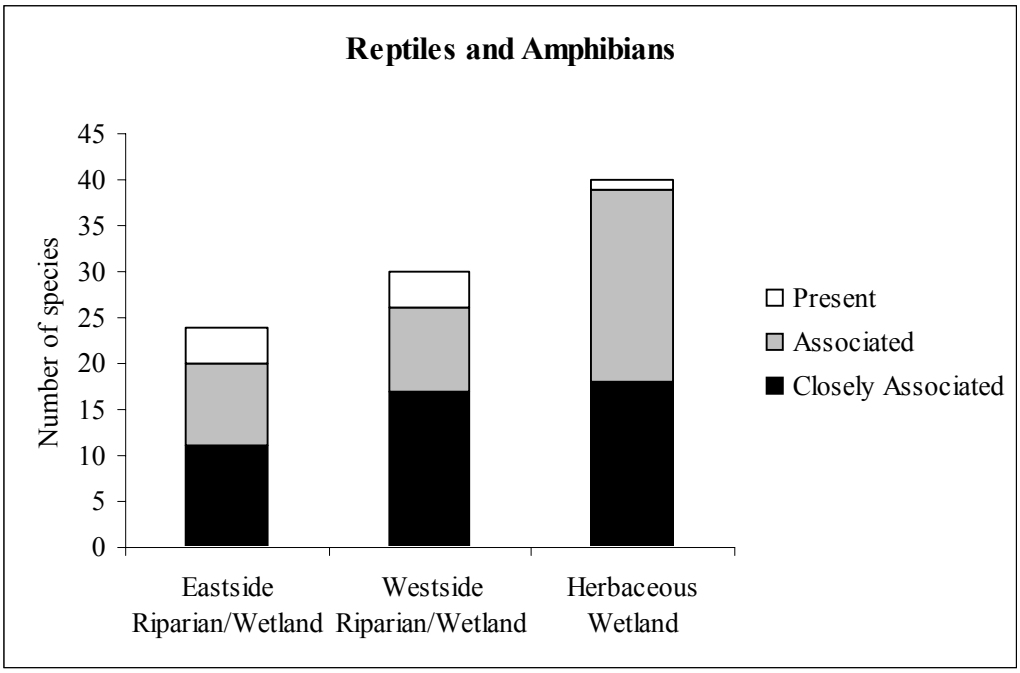

Figure 2-12. The number of reptile and amphibian species found in riparian and herbaceous wetlands in Washington and Oregon (from Kauffman et al. 2001). Note: the authors do not define "present," "associated," or "closely associated."

Birds

Overall, 266 (72%) of the 367 species of birds in Oregon and Washington use freshwater, riparian, and wetland habitats. More striking, 204 (77%) of the 266 species of inland birds that breed in the two states do so in riparian and wetland environments (Kauffman et al. 2001). Figure 2-13 shows how many bird species use the riparian and herbaceous wetlands in the region.

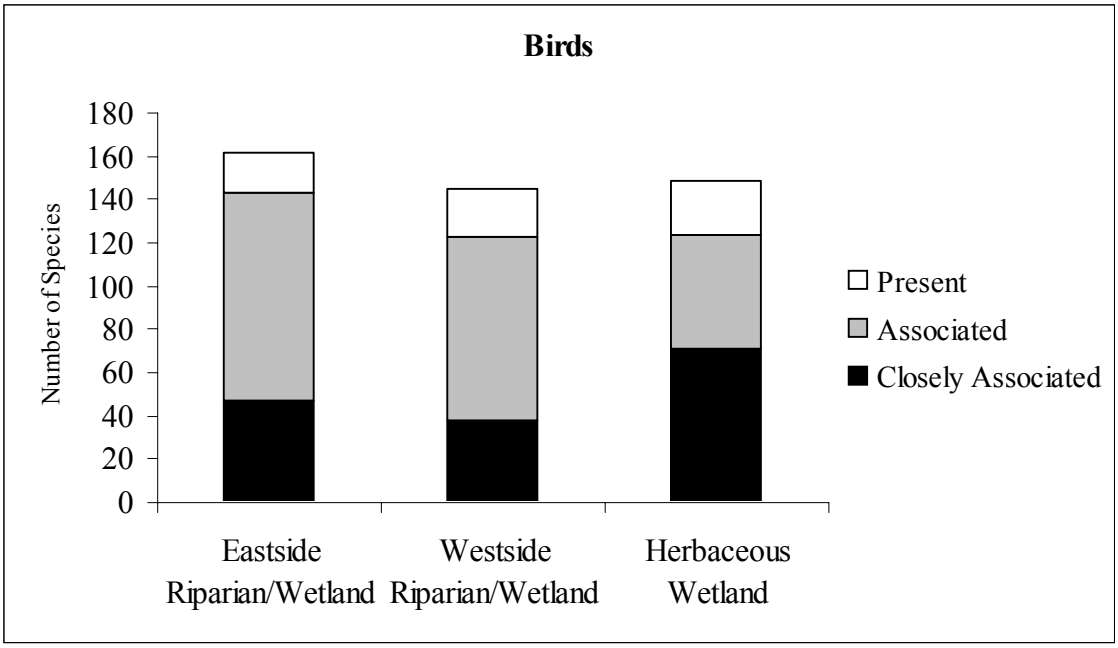

Figure 2-13. The number of bird species found in riparian and herbaceous wetlands in Washington and Oregon (from Kauffman et al. 2001).

Mammals

Ninety-five of the 147 mammal species (65%) in the two states use the riparian/wetland ecosystem (Kauffman et al. 2001). All the "furbearers" (e.g., mink, otter, beaver, raccoon, etc.) use this ecosystem, and all but one of the big game animals (deer, elk, moose, etc. with the exception of bighorn sheep) rely on these areas for part of their habitat requirements. Figure 2-14 shows the number and degree of association of mammals to the three types of wetland habitats considered in Kauffman et al. (2001).

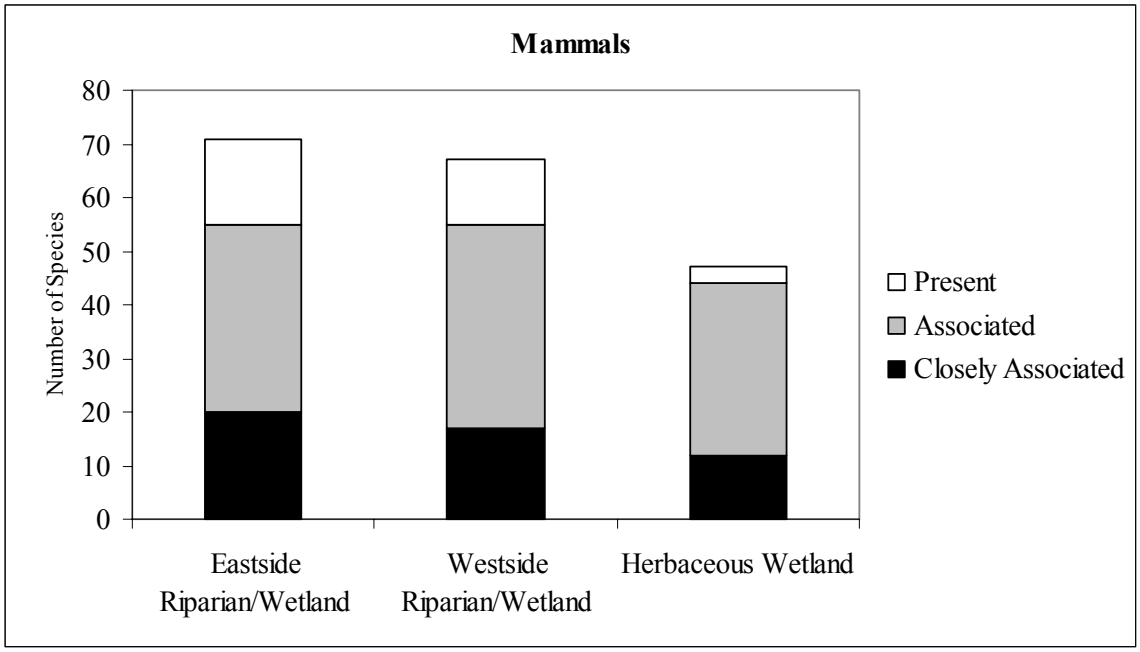

Figure 2-14. The number of mammal species found in riparian and herbaceous wetlands in Washington and Oregon (from Kauffman et al. 2001).

2.5.3.3 Habitat Requirements of Some Wetland-Dependent Species in Washington

Invertebrates

Invertebrates have evolved unique adaptations enabling them to occupy most wetland habitats and most parts of the food web. Consequently, wetland invertebrates are considered pivotal components of the food webs in wetlands (Mitsch and Gosselink 2000). As filter feeders, shredders, and scrapers, insects convert microorganisms and vegetation into biomass, providing significant food that then becomes available to animals higher in the food web (secondary and tertiary consumers). Research focusing on aquatic invertebrates in wetlands indicates the importance of invertebrates in energy and nutrient transfer within aquatic ecosystems (Rosenberg and Danks 1987).

The abundance of invertebrates in wetlands can be extremely large. Leeper and Taylor (1998) measured densities in excess of 700,000 organisms per square meter in shallow depressional wetlands of South Carolina.

Factors found to influence the distribution, richness, and abundance of invertebrates in wetlands include:

- Species richness of invertebrates is high in **water interspersed with stands of emergent vegetation** (Voigts 1976).

- **Decaying wood** provides an important habitat for invertebrates (Maser et al. 1988).
- A **mix of plant assemblages** exhibits greater richness of invertebrate species than a single assemblage (Andrews and Hasler 1943, Dvorak and Best 1982, Lodge 1985, Balla and Davis 1995). Furthermore, the density of invertebrates varies considerably among species of submerged aquatic plants (Murkin and Batt 1987), and different invertebrate species are found on different plant species (Cyr and Downing 1988).
- **Permanent flowing water** is a habitat feature that supports a unique assemblage of invertebrate species (Needham and Needham 1962, Wiggins et al. 1980, Rolauffs et al. 2001). Furthermore, the invertebrates in flowing permanent channels are an important resource for many other aquatic species such as fish (Needham and Needham 1962).
- Higher invertebrate species richness was found in wetlands that underwent marked **seasonal changes in water regime** compared to those that contained permanent water with little water level fluctuation (Balla and Davis 1995).
- **Vegetation** is a major factor shaping wetland invertebrate communities (Krieger 1992, Wissinger et al. 1999).

Not much is known about invertebrate distributions in different soil surfaces within a wetland. However, data from rivers, streams, and lakes show that the local invertebrate species have preferences for specific surfaces (Gorman and Karr 1978, Dougherty and Morgan 1991). In streams it is well known that the composition of midges (chironomids) is strongly affected by characteristics of the sediment surface (McGarrigle 1980, Minshall 1984).

Amphibians

Amphibians are a vertebrate group that, in the Pacific Northwest, includes wetland-breeding frogs and salamanders. Both the richness and abundance of amphibians in wetlands indicate that they are important in wetland food webs (Hruby et al. 1999). Some native species only breed for a short time in wetlands and then live in uplands as adults. Other species are found in or close to wetlands throughout the year. However, the eggs and larvae of all wetland-breeding species require water for development (Hruby et al. 1999).

Other information known about amphibians in wetlands includes the following:

- The **presence of corridors** leading to other wetlands or to upland habitat is critical. Relatively undisturbed migration routes between a wetland and upland feeding and hibernation sites are important for many amphibian species (Heusser 1968, Berven and Grudzien 1990, Beebee 1996). Moreover, dispersal routes for recolonization are critical when populations are eliminated by random processes including drought (Pounds and Crump 1994), disease (Bradford 1991), or

pollution (Richter, personal communications), or when populations produce insufficient offspring to permanently occupy a site (Gill 1978a, 1978b, Sinsch 1992). Finally, inbreeding is minimized when the amphibians within a wetland are members of a population that extends across several wetlands (Sofgren 1991, 1994, Pechmann and Wilbur 1994).

- **Conditions in the buffers** of a wetland are especially important in providing cover to amphibian females and to newly metamorphosed animals. Female red-legged frogs (*Rana aurora*), Northwestern salamanders (*Ambystoma gracile*) (Richter, personal communication), and long-toed salamanders (*A. macrodactylum*) (Beneski et al. 1986, Leonard and Richter 1994) generally wait in buffers near wetlands until environmental and biological conditions are favorable to spawning. They then enter wetlands during one or a few nights to spawn, thereafter quickly retreating to the cover provided by buffers. Buffers are important to the tiger salamander (*A. tigrinum*) seeking shelter in rodent burrows during the first days following emigration from ponds in which they are born (Loredo et al. 1996).

- Most species of amphibians generally avoid both exposed water and densely vegetated sites, instead selecting habitats with an interspersion of both features (Strijbosch 1979, Ildos and Ancona 1994). Research suggests that most species of amphibians select areas in which **exposed water and vegetation are interspersed** for laying eggs (Richter, personal communication).

- Water level fluctuations are known to have a significant influence on amphibians (Richter 1996, 1997). Most species of amphibians in temperate climates minimize exposure of eggs to fluctuating depths and temperatures by both spawning at mid-depth and by submerging eggs below the surface (Richter 1997). Amphibian egg development also depends on permanent or partial submergence, and therefore optimum habitat conditions are those where **water levels are stable** from spawning through hatching. In most Puget Sound species this is from mid-December through mid-May. Although mean water level fluctuations exceeding approximately 8 inches (20 cm) have been correlated to decreased amphibian richness in wetlands (Azous and Richter 1995), their experiments suggest that extended drops of more than approximately 3 inches (7 cm) from the time of egg laying through hatching may harm the Northwestern salamander.

- Experimental evidence suggests that **vegetation structure**, particularly plant shape and stem diameter, rather than the species of the plant, is most important to salamanders. Wetland surveys and controlled field studies of several Northwest salamanders confirm that distinct stem widths are preferred (Richter 1997).

Anadromous Fish

Anadromous fish are those that spend all or part of their adult lives in salt water and return to freshwater streams and rivers to spawn. There are 12 species of anadromous fish in the Pacific Northwest (PSMFC 2001), but not all are regular users of wetlands.

The Pacific Northwest salmonids (species of the genus *Oncorhynchus*) have recently been the focus of much research because of the status of some species as threatened or endangered. The most common anadromous species that uses wetlands is the coho salmon (*Oncorhynchus kisutch*). Other anadromous fish noted in wetlands found in side channels, or old oxbows, of rivers and streams (off-channel wetlands) include cutthroat trout (*O. clarki*) and steelhead (*O. mykiss*) (Peterson 1982). It is not the intent of this review to summarize all the information available on the habitat needs of salmonids. Some of the most important habitat structures in wetlands that have been found to be important for anadromous fish are summarized below:

- "Slope" and "flats" wetlands in Washington are the only classes of wetlands that do not have the potential to provide habitat for anadromous fish because, by definition (Brinson 1993a) they do not have **ponded or impounded surface water**.
- A wetland must have a **surface water connection** to a salmon-bearing stream or river if fish are to enter or exit the wetland (Hruby et al. 1999).
- **Interspersion between land and water** in a wetland is important because the contact zones between exposed water and vegetation provide protection from wind, waves, and predators, and may provide natural territorial boundaries (Golet and Larson 1974).
- Anadromous fish need a certain **water depth** for optimum habitat conditions. Narver (1978) observed juvenile coho moving into areas with water depth over approximately 18 inches (45 cm) and lower velocities (6 inches [15 cm] per second) when temperatures decline below approximately 41°F (7°C). Beaver ponds and off-channel areas with similar depths also provide habitat (Reeves et al. 1989). Survival and growth of overwintering fish may be maximized in systems that contain both shallow pools and deeper ones (Peterson 1982).
- **Overhanging vegetation** provides both temperature control and protection from predation. McMahon (1983) reported the need for streamside vegetation for shading. Small coho juveniles tend to be harassed, chased, and nipped by larger juveniles unless they stay near the bottom, obscured by rocks or logs (Groot and Margolis 1991). Cover for salmonids in wetlands can be provided by (Giger 1973):
  - Overhanging vegetation
  - Submerged vegetation
  - Submerged objects such as logs and rocks
  - Floating debris
  - Deep water
  - Turbulence

– Turbidity (the assumption seems to be that cloudy water reduces the visibility of fish in open water where birds may prey on them)

Resident Fish

Fish that do not migrate out of wetlands are considered "resident fish." Many different fish species use wetlands and it is not practical to list all that occur in Washington's wetlands.

Before the late 1800s, the only resident freshwater game fish living in Washington State were trout, char, whitefish, burbot, and squawfish. Since then there has been a widespread and often indiscriminate introduction of game species from other parts of the nation (WDFW 1999).

Some of the characteristics in wetlands that provide habitat for resident fish include:

- Resident fish, like anadromous fish, need a **range of water depths** for different parts of their life cycles (Hruby et al. 1999). Shallow waters provide refuge for young fish, while the deeper waters provide refuge for the larger adults. Varying water depths also provide different potential food sources since they are host to different populations of plants and invertebrates. (See the earlier discussion of invertebrate habitat.)

- **Shorelines between exposed water and vegetation** provide protection from wind, waves, and predators, and may provide natural territorial boundaries (Golet and Larson 1974).

- **Overhanging vegetation** provides both temperature control and protection from predation (McMahon 1983).

- **Large woody debris** plays an important role in the Pacific Northwest, creating and enhancing fish habitat (Bisson et al. 1987).

Wetland-Dependent (or Associated) Birds

Wetland-dependent bird species are those that depend on aspects of the wetland ecosystem for some part of their life needs: food, shelter, breeding, or resting. Kauffman et al. (2001) reviewed the literature and found a very high richness and abundance of birds in wetland ecosystems of the Pacific Northwest. They found that:

> *All 23 species of waterfowl that breed regularly in the western U.S. south of Alaska do so in riparian and wetland environments. Similarly, all 14 western species of waders, a group consisting of cranes, rails, herons, and ibises, depend on riparian and wetland habitats for most of their life cycles. Shorebirds, which include stilts and avocets, sandpipers, and plovers are typically dependent on freshwater, riparian, and wetland habitats. Interior wetlands (i.e. east of the Cascades) also provide crucial stopover habitat for 37 species during migration.*

A review of the specific habitat requirements of all birds using wetlands is beyond the scope of this document. General characteristics of wetlands that provide good habitat for wetland-dependent birds include the following:

- The **condition of the wetland buffer** is an important characteristic for bird habitat. Trees and shrubs provide screening for birds, as well as providing additional habitat in the buffer itself (Johnson and Jones 1977, Milligan 1985).
- The **size of the buffer** as well as its condition is important. Milligan (1985) concluded that bird species richness in urban wetlands of King County was strongly correlated with a buffer of at least 50 feet (15 m).
- **Snags** are a source of cavities and perches for wetland-associated birds. Several species of birds use already existing cavities for nesting and/or refuge locations. Dead wood attracts invertebrates and other organisms of decay, which in turn provide a food source for many species of birds (Davis et al. 1983).
- Some bird species may require **several habitat types** such as open water and grasslands in close proximity to aid their movements from one type to another (Hunter 1996, Gibbs et al. 1991).
- **Embayments and peninsulas** in a wetland with open water provide "micro-habitats" for certain species that require hiding cover or those seeking security within a more enclosed system (USDI 1978).
- The **proximity of a wetland to open water or large fields** increases its utility to migrant and wintering waterfowl. If there is strong connectivity between relatively undisturbed aquatic areas, the suitability of a wetland as waterfowl habitat increases (Gibbs et al. 1991).
- **Open water of varying depths** provides greater diversity of foraging habitat for a greater variety of water birds (USDI 1978).
- A **full canopy can limit access** to open water in a wetland because birds have difficulty flying in and out. This may be best illustrated by great blue herons (*Ardea herodias*), which will be reluctant to fly down to a body of water if the tree canopy above is totally closed because rapid escape may be difficult or impossible (USDI 1978).

Wetland-Dependent (or Associated) Mammals

For the purpose of this review it is not practical to synthesize the specific habitat requirements of all mammal species using wetlands. The richness of mammal species using wetlands can be very high. Kauffman et al. (2001) report that 79 mammal species east of the Cascades and 69 on the west side use riparian wetlands. The wetlands associated with stream corridors characteristically have greater species richness than upland sites and provide habitat for some species that are not found elsewhere. About

half of the species using riparian wetlands in the Pacific Northwest breed and feed in them (Kauffman et al. 2001.)

Many mammals use wetlands but do not depend on them for some part of their life needs (food, shelter, breeding, or resting). These mammals are considered to be wetland users rather than wetland-dependent. The following bullets summarize some general information about the characteristics of wetlands that provide good habitat for four mammal species usually considered to be wetland-dependent in Washington State (Hruby et al. 1999). These species include the beaver (*Castor canadensis*), muskrat (*Ondatra zibethicus*), river otter (*Lutra canadensis*), and mink (*Mustela vison*).

- Wetlands with a **relatively undisturbed buffer** are important to these four species (and others) because the buffers:
  - Minimize disturbance (Allen and Hoffman 1984, Burgess 1978)
  - Provide habitat for prey species and food sources for mammals (Allen 1983, Dunstone 1978, Brenner 1962)
  - Provide cover from predators (Melquist et al. 1981)
  - Allow den sites for resting and reproduction (Allen 1983)
- Beavers prefer a **seasonally stable water level** (Slough and Sadleir 1977). Large fluctuations in water levels may also affect the suitability of a wetland for muskrats (Errington 1963). Wetlands subject to heavy spring runoff or flash floods that rapidly raise the water level may cause flooding of burrows (Errington 1963).
- For beavers, **water depth must be sufficient** to accommodate lodges and bank dens and to allow free movement from the lodge to food caches during the winter. For example, freezing of the food cache is a limiting factor on beaver and muskrat survival in the Columbia Basin (Tabor, personal communication). Freezing of a pond to the bottom can be disastrous to muskrat populations (Schmitke 1971). Deep water will also provide protection from predators (Easter-Pilcher 1987). In the Columbia Basin beavers and muskrats need at least 4 feet (1.3 m) of permanent water to allow access to food caches during the winter when the surface is frozen (Hruby et al. 2000).
- River otters feed primarily on fish (Kauffman et al. 2001). Wetlands with **fish**, therefore, provide better otter habitat than those without.
- **Vegetated corridors** leading to and from wetlands are considered an important feature in assessing the suitability as habitat (Hruby et al. 2000). Dispersal is a fundamental process in regulating populations among these and other mammals (Kauffman et al. 2001).
- Muskrats and beavers use **persistent emergent cover** for security and feeding (Errington 1963). Allen (1983) believes that beavers prefer herbaceous vegetation over woody vegetation during all seasons, if available.

- **Interspersion of vegetation and open water** is an important characteristic of wetlands as habitat for mammals. High interspersion rates increase the abundance of prey for mink and river otter (i.e., muskrats, water birds, fish) (King 1983). Food abundance and availability appeared to have the greatest influence on habitat use by river otter in Idaho (Melquist and Hornocker 1983). Classic muskrat studies by Dozier (1953) and Errington (1937) indicate that optimum muskrat habitat is 66 to 80% of the wetland in emergent vegetation with the remainder in open water.

2.5.3.4 Habitat for Plants

Relatively few plant species of the thousands on Earth have adapted to the harsh conditions in wetlands. Major stressors are lack of oxygen, salt, and water level fluctuations in an environment that is neither fully aquatic nor terrestrial (Mitsch and Gosselink 2000). These strong selective pressures have produced a group of plant species that is unique to wetlands and whose maintenance has become an issue in regional biodiversity (Gibbs 2000).

All wetlands provide the four basic requirements for plant growth (space, water, light, and nutrients) to some degree. Differences can be found among wetlands in the number of plant species they contain. Recent research has been focused on the characteristics of wetlands that affect plant richness, as summarized below:

- **Specific water regimes**, such as permanent inundation, seasonal flooding, or saturation, result in unique plant communities (Mitsch and Gosselink 2000).

- The **duration of individual flooding events** is important in separating plant communities because the duration affects germination of seeds in different ways (Casanova and Brock 2000).

- The **water regime** in a wetland can either limit the number of species present or enhance it, depending on types of water level fluctuations and physical energy of the water regime (Mitsch and Gosselink 2000).

- Plant richness in a wetland generally follows the ecological theory that maximum richness occurs at **intermediate levels of environmental stress** (Johnson and Leopold 1994). For example, water level fluctuation is an environmental stress (Mitsch and Gosselink 2000). Wetlands with large water level fluctuations, therefore, would be expected to have fewer plant species than those with moderate water level fluctuations. On the other hand, wetlands with very small water level fluctuations (low stress) would also be expected to have fewer plant species.

- Observations show that the distribution of species within a wetland is primarily a function of water depths (Spence 1982 in Van der Valk et al. 1994). As a result, wetlands with a **range of water depths** tend to have higher richness than those with fewer (Hruby et al. 1999).

2.5.3.5 Supporting Food Webs (Primary Production and Export)

Wetlands are known for their high primary productivity (production of plant material) and the subsequent export of this organic matter to adjacent aquatic ecosystems. The exported organic matter provides an important source of food for most downstream aquatic ecosystems (Mitsch and Gosselink 2000).

Plant material produced in wetlands breaks down into smaller and smaller particles and becomes increasingly nutritious due to the activity of bacteria, fungi, and protozoa (Sipple 2002). This decomposed plant material, including the various microbes that colonize it, feeds many small aquatic invertebrates and small fish. These invertebrates and fish then serve as food for larger predatory amphibians, reptiles, fish, birds, and mammals (Sipple 2002).

The following summarizes general characteristics of wetlands that have high production and provide excellent support for aquatic food webs:

- In general, wetlands **where water flows through the system** have higher levels of primary production and export than those where water is impounded without leaving (Mitsch and Gosselink 2000).
- The **water level fluctuation** as well as movement of water mentioned above through the wetland and its **soils** is one of the most important determinants of primary productivity (Mitsch and Gosselink 2000).
- Performance of this function requires both that **organic material** is produced and that a mechanism is available to move the organic matter to adjacent or contiguous aquatic ecosystems (Hruby et al. 1999).

2.5.4 Summary of Key Points

- The residence time of water in the wetland and filtering by wetland vegetation are major processes influencing removal of sediments, phosphorus, and toxics from surface water. Wetland vegetation typically removes 80 to 90% of sediment from runoff. Wetlands with seasonal inundation or saturation have conditions that promote removal of nitrogen from surface runoff. In order for a wetland to provide functions that improve water quality, however, surface water containing pollutants must first enter the wetland.
- The capacity of a wetland to store surface water affects its ability to reduce peak flows, as do the amount of flow from the upper watershed that enters the wetland and the amount of woody vegetation present. Reducing peak flows helps to decrease downstream erosion.
- Only wetland types that impound surface water have the potential to provide groundwater recharge.

- Wildlife species can be "wetland dependent" or "wetland users." Wetland-dependent species (such as amphibians) require a wetland for at least part of their life cycles. Wetland users (such as deer) come to wetlands for such needs as water or cover.
- The characteristics of wetlands that provide habitat depend on species and life stage. Characteristics that are important for many species include vegetation structure, water depth, water level fluctuation, buffers, snags, and connections to other habitats in the landscape.
- Wetlands have high productivity of plant material. Decomposed plant material can be exported downstream, providing food for insects, fish, and other organisms in the food web.

2.6 Chapter Summary and Conclusions

The functions of wetlands are things that wetlands "do." They represent the many interactions possible among the different components of the ecosystem found in wetlands. There are many interactions that occur in wetlands and they occur at many scales. In general, however, functions are grouped into three broad categories: 1) biogeochemical interactions, 2) hydrologic interactions, and 3) interactions that maintain food webs and habitats for plants and animals.

The primary factors that control wetland function are climate, geomorphology, the source of water, and the movement of water. These factors affect wetland functions directly or through a series of secondary factors including nutrients, salts, toxic contaminants, soils, temperature, and the connections created between different ecosystems. The factors that control wetland functions interact with each other and there are many feedback loops. A number of conceptual models have been developed to help visualize and understand the complexity of the interactions between environmental factors, environmental processes, and wetland function.

The major environmental factors of geomorphology, source of water, and the movement of water are the basic characteristics used to classify wetlands in Washington into groups of wetlands that have similar functions and can be expected to perform these functions in similar ways. Freshwater wetlands in Washington are divided, based on how they function, into two domains, five regions, and six classes.

The environmental factors that control the structure and functions of a wetland occur at both the landscape scale and the site scale. For example, riverine wetlands will be affected to a great degree by processes operating at the scale of the entire watershed of the river, whereas depressional wetlands will be subject to processes that occur only within the basin that contributes surface or groundwater to the wetland.

The most important factors that control functions at an individual site may occur somewhere else in the watershed. Information about factors that control functions at the larger scale is still evolving. The importance of the environmental factors that occur at

the larger, landscape scale, however, should not be minimized for lack of information. Ongoing research is continually strengthening our understanding of these critical factors.

The links between wetland functions and the landscape have been well described by the National Academy of Sciences (National Research Council 1995):

> *Individual wetlands function to a large degree through interaction with the adjacent portions of the landscape and with other wetlands. For example wetlands whose principal source of water is groundwater depend on that water infiltrating in the surrounding uplands. If these uplands are paved, clear-cut, or farmed, the amount of water recharge is significantly reduced and the wetland may dry up or become smaller. No single wetland or aquatic site could support anadromous fish. The connections between individual wetlands, aquatic systems, and terrestrial systems are critical to the support of many species. Furthermore, flood control and pollution control are determined by the number, position, and extent of wetlands within watersheds. Thus, the landscape gives proper context for the understanding of some wetland functions.*

An understanding of wetland functions for the purposes of managing and protecting them will require knowledge of how the major controls of functions change or are impacted by humans at all scales. We need to understand how climate, topography, and the movement of water, nutrients, sediment, etc. are affected by human activities in the entire watershed, as well as in the immediate vicinity of the wetland. Chapter 3 describes the environmental disturbances caused by different human uses of the land. Chapter 4 then carries this information forward to discuss how the disturbances caused by human activities affect specific functions of wetlands.

Chapter 3
Environmental Disturbances Caused by Human Activities and Uses of the Land

3.1 Reader's Guide to this Chapter

In Chapter 3, the discussion shifts from wetland functions and the environmental factors that control the performance of functions to the major disturbances created by human activities and uses of the land and water. In this context, a *disturbance* is an event that changes an environmental factor that controls wetland functions.

3.1.1 Chapter Contents

Major sections of this chapter and the topics they cover include:

Section 3.2, Introduction to Human-Caused Disturbances at the Landscape Scale provides an overview of how human land uses change the dynamics and structure of the ecosystems in watersheds by creating various types of disturbances. The section provides a general overview of how human activities affect the movement and quality of water and connections between habitats across the landscape. The thresholds at which the amount of impervious surface in a watershed begins to impact aquatic systems are also discussed.

The chapter continues with separate sections for four major types of human land uses in Washington State (agriculture, urbanization, logging, and mining) and how they create disturbances. The different uses of the land by humans are divided into these four categories because most of the literature found discusses the disturbances and impacts of human activities in these terms.

Each major land use is addressed in a separate section, as follows:

Section 3.3, Disturbances Caused by Agriculture discusses the changes in the input of nutrients, salt, sediment and contaminants caused by agriculture.

Section 3.4, Disturbances Caused by Urbanization discusses the changes urbanization has caused to the water regime in watersheds and how this land use has resulted in sedimentation, nutrients, input of contaminants, and fragmentation of habitat.

Section 3.5, Disturbances Caused by Logging refers the reader, after a brief summary, to another synthesis that summarizes the literature on the disturbances created by logging.

Section 3.6, Disturbances Caused by Mining discusses the increased level of heavy metals and acidity in surface waters that results from mining.

For each of these four land uses, several types of disturbances that change the factors controlling wetland functions are discussed, including (as applicable):

- Changing the physical structure within a wetland (e.g., filling, removing vegetation, tilling soils, compacting soils)
- Changing the amount of water (increasing or decreasing the amount)
- Changing the fluctuation of water levels (frequency, amplitude, direction of flow)
- Changing the amount of sediment (increasing or decreasing the amount)
- Increasing the amount of nutrients
- Increasing the amount of toxic contaminants
- Changing the acidity (acidification)
- Increasing the concentration of salt (salinization)
- Decreasing the connection between habitats (fragmentation)
- Other disturbances (noise, etc.)

Section 3.7, Chapter Summary and Conclusions ties together the major concepts presented in the chapter.

3.1.2 Where to Find Summary Information and Conclusions

Each major section of this chapter concludes with a brief summary of the major points resulting from the literature review on that topic in a bullet list format. The reader is encouraged to remember that a review of the entire section preceding the summary is necessary for an in-depth understanding of the topic.

For summaries of the information presented in this chapter, see the following sections:

- Section 3.2.4
- Section 3.3.11
- Section 3.4.9

In addition, Section 3.7 provides a summary and conclusions about the overarching themes gleaned from the literature and presented in this chapter.

3.1.3 Data Sources and Data Gaps

There are abundant data on some of the subjects related to the effects of land uses on water quantity, water quality, and some habitat issues. For example, the Puget Sound Wetlands and Stormwater Management Research Program (Azous and Horner 2001) was one important source of scientific literature on how changes in watershed land uses affect the physical, chemical, and biological factors that control wetland functions in the lowlands of Puget Sound.

In contrast, literature on the effects of agricultural practices in the Pacific Northwest is limited. Most studies originate from the prairie pothole region of the United States, the high mountain West, or California. Literature related to agriculture from outside the Pacific Northwest region has been included in this synthesis when it was judged to be relevant to Washington.

No scientific research was found that examined the question of whether some wetlands in eastern Washington existed before the onset of irrigation projects. Research conducted by Adamus on irrigated agricultural lands from the high basin country of Colorado (Adamus 1993) may not be germane to eastern Washington, but it is included in the section on agriculture as a source of discussion on the influence of irrigation on wetlands.

3.2 Introduction to Human-Caused Disturbances at the Landscape Scale

3.2.1 The Link between Wetland Functions, Human Land Uses, and Changes in Wetland Ecosystems

Human activities and human uses of the land change the structure and dynamics of ecosystems (Dale et al. 2000). The disturbances caused by human land uses create a "stress" on the ecosystem to which it responds. The changes in the ecosystem are then reflected in changes to how it functions. Scientists sometimes use the term "stressor" to distinguish those disturbances that have a significant impact on an ecosystem from those that have little impact (see for example Adamus et al. 2001, Laursen et al. 2002). For the purposes of this discussion, however, the term "disturbances" is used for clarity. All the disturbances discussed herein are considered to have significant impacts on ecosystems.

In terms of wetlands and their functions, a disturbance can be considered as a condition or event that changes one of the environmental factors that control wetland functions. For example, nutrients are a factor that controls wetland functions. If nutrients from residential lawns flow to a depressional wetland that has limited nutrients, such as a bog, this excess can change the dominant plants in the bog and its habitat structure. In this case, the addition of nutrients that are in excess of those found in the absence of human activities is a disturbance on the ecosystem of the wetland.

This example illustrates how land use changes can influence large-scale ecosystem processes, resulting in disturbances to the factors that control wetland functions. It also illustrates how the topics discussed in Chapters 2, 3, and 4 of this volume are linked:

- The movement of nutrients throughout a basin, as described in the example, is one of several environmental factors that control wetland functions. These factors and the way in which they control wetland functions are the subject of Chapter 2.
- The maintenance of residential lawns is an example of a human activity that effect environmental factors that control wetland functions. The application of excess nutrients (fertilizer) in the example creates a disturbance when the nutrients flow from the lawn into a bog. This chapter (Chapter 3) describes how different kinds of human activities and uses of the land create environmental disturbances.
- When the excess nutrients reach the bog, they cause a change in its plant community and its habitat structure because the plant communities are adapted to a low-nutrient, acidic environment. Chapter 4 describes how disturbances caused by human land uses result in changes to wetland functions.

Figure 3-1 reviews the connection between the controls of wetland functions, human disturbances, and the functions of wetlands.

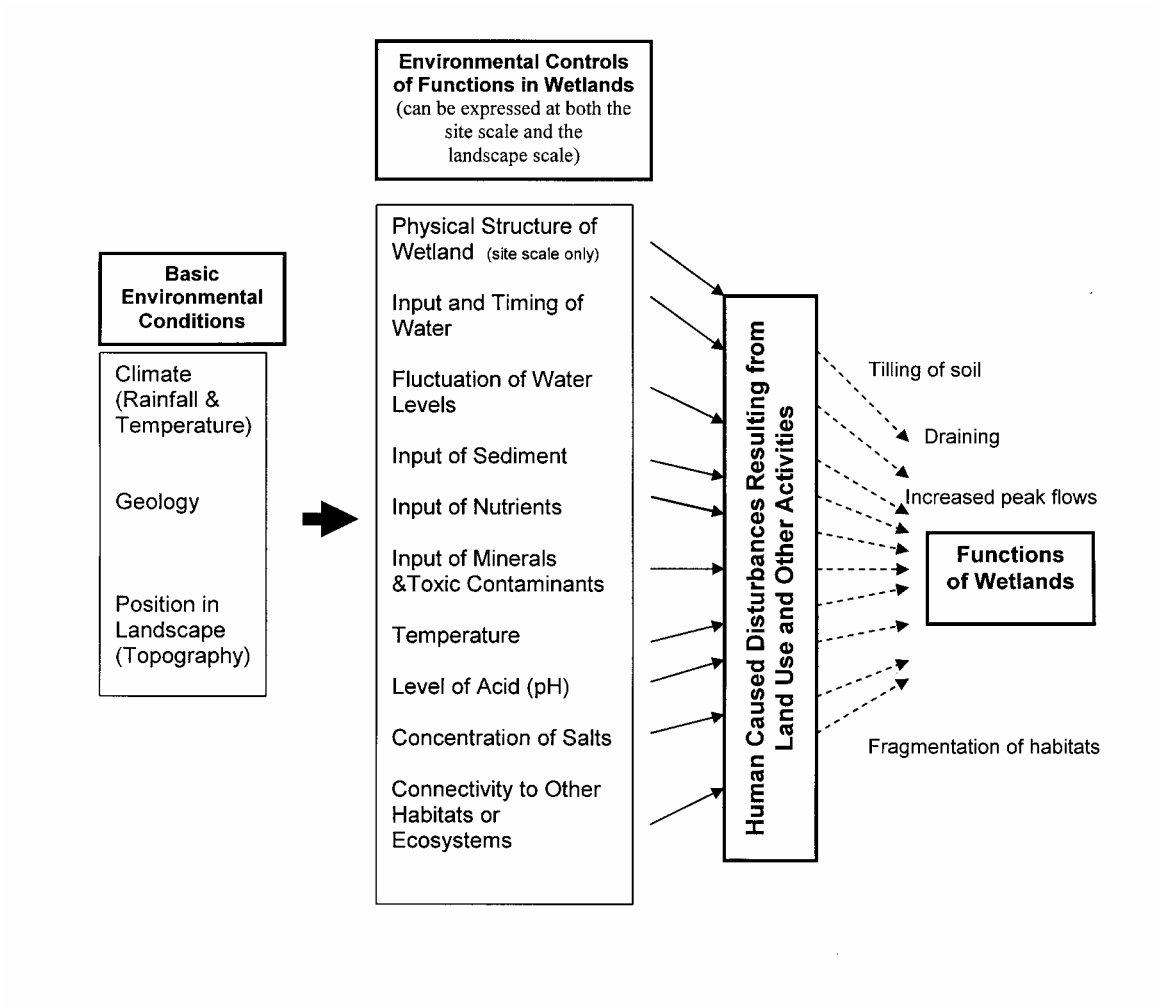

Figure 3-1. Diagram summarizing the environmental factors that control functions of wetlands and how they interact with human disturbances. The basic environmental conditions establish and determine the factors that control the functions of wetlands. The controls can occur at both the landscape and site scales. Human activities cause disturbances that affect these controls in many different ways and thereby alter the performance of wetland functions. The figure gives some examples of the disturbances.

3.2.2 Types of Disturbances Resulting from Human Land Uses

Many different types of disturbances have been identified in the literature. For the purposes of organizing the information in this chapter, the list developed by Adamus et al. (2001) and shown in Table 3-1 is used because it was developed specifically to address impacts to wetland functions. Table 3-1 lists the types of disturbances that can impact wetlands and the scale at which the disturbances occur. Many disturbances that result from human land uses can occur over large areas such as basins and sub-basins (called the landscape scale), as well as in the immediate vicinity of the wetland itself (called the site scale).

Table 3-1. Summary of human disturbances and the scale at which they can occur.

Disturbance	Scale of Disturbance
Changing the physical structure within a wetland (e.g., filling, removing vegetation, tilling soils, compacting soils)	Site
Changing the amount of water (increasing or decreasing the amount)	Landscape and site
Changing the fluctuation of water levels (frequency, amplitude, direction of flow)	Landscape and site
Changing the amount of sediment (increasing or decreasing the amount)	Landscape and site
Increasing the amount of nutrients	Landscape and site
Increasing the amount of toxic contaminants	Landscape and site
Changing the temperature	Mostly site
Changing the acidity (acidification)	Landscape and site
Increasing the concentration of salt (salinization)	Mostly site
Decreasing the connection between habitats (fragmentation)	Landscape
Other disturbances (noise, etc.)	Landscape and site
This table is a synthesis of the information presented by Adamus et al. (2001) and in the literature review done for this report.	

3.2.3 General Disturbances to the Movement of Water at the Landscape Scale

The movement and sources of water in a watershed are two critical factors controlling how wetlands function. Many human land uses change the movement and sources of water in a watershed, thereby creating a disturbance that affects the performance of functions in wetlands. The following provides some background on how human activities result in disturbances to the movement and sources of water.

The literature is quite clear that the frequency, timing, and duration of water in the landscape determine the presence of a wetland and the functions that it provides (see Chapter 2). How water enters a wetland, how long it is present, and the depths to which it is impounded all influence the functions that a wetland can provide or perform (Brinson 1993a, Mitsch and Gosselink 2000). Water moves or carries sediment, nutrients, woody debris, and energy throughout the watershed (Naiman et al. 1992).

Surface and subsurface water flows through the landscape within *drainage systems*. These drainage systems are often called basins, sub-basins, watersheds, or river basins depending on the size of the area. In this document, drainage systems are generally referred to using one of two terms:

- *Watersheds:* A watershed is a geographic area of land bounded by topographic high points in which water drains to a common destination.
- *Contributing basins:* A contributing basin is an area from which water drains to a particular wetland.

Booth (1991) succinctly summarizes the concept of a drainage system as follows:

> *Drainage systems consist of all of the elements of the landscape through which or over which water travels. These elements include the soils and the vegetation that grows on it, the geologic materials underlying that soil, the stream channels that carry water on the surface, and the zones where water is held in the soil and moves beneath the surface. Also included are any constructed elements, including pipes and culverts, cleared and compacted land surfaces, pavement and other impervious surfaces that are not able to absorb water at all.*

The movement and routing of water flows above and below the surface is the primary force in transporting nutrients, sediment, salts, and contaminants, and this in turn affects the functions provided by wetlands (Naiman et al. 1992). Changes in the amount of water, as well as in the frequency and fluctuations of water volumes, can alter how sediments, nutrients, and toxic contaminants come into a wetland. Changes in the movement of water resulting from human activities at the landscape scale can therefore have severe impacts on wetland functions throughout a watershed.

The following subsections provide background on how water moves in undisturbed landscapes as well as those that have been changed by human activities. The purpose of this discussion is to provide a context for understanding how human activities and uses of the land create the disturbances discussed later in the chapter.

Terms used to describe water regimes: hydrology vs. hydroperiod vs. hydrologic

Hydrology and hydroperiod are often used interchangeably to mean how water moves. *Hydrology*, as defined by Webster, is "the scientific study of the properties, distribution, and effects of water in the atmosphere, on the earth's surface, and in soil and rocks." The term hydrology means the study of how water moves.

Hydroperiod (not defined by Webster) is commonly used to refer more precisely to the periodicity of water; the timing (seasonal or otherwise) and duration of water's presence or absence within a particular aquatic feature, such as a wetland. It is "the seasonal occurrence of flooding and/or soil saturation, encompassing the depth, frequency, duration and seasonal pattern of inundation." (Azous et al. 2001). Mitsch and Gosselink (2000) define hydroperiod as "the seasonal pattern of the water level of a wetland . . . a hydrologic signature of each wetland type." Hydroperiod, in this context, refers to seasonal changes in wetland water level conditions caused by regular annual changes in water availability. This should be differentiated from the water level fluctuations driven by single or serial storm events.

Hydrologic is an adjective derived from the word "hydrology." It refers to the properties, distribution, and effects of water. Thus a term such as "hydrologic processes" refers to the environmental processes that involve the properties, distribution, and effects of water.

For this report, "hydroperiod" is used to refer to the pattern of water movement in a particular wetland or type of wetland. The term "hydrology" has been retained when direct quotes from sources use that term even if it has been misused. "Hydrologic" is used when an adjective is needed to describe the patterns of water movement.

3.2.3.1 Movement of Water in Undisturbed Landscapes

In undisturbed conditions, very little of the precipitation falling on the ground ends up in surface runoff, even in areas of high annual rainfall such as the Pacific Northwest. Areas with natural vegetation provide high rates of interception, infiltration, and evapotranspiration (Ziemer 1998). The water either drips off leaves to the soil below; flows down the stems, leaves, and bark to the soil; or evaporates into the air, never reaching the ground.

Water that infiltrates into permeable surfaces either moves downgradient as shallow groundwater, infiltrates into a deeper water table, or is taken up by plant roots and rerouted back into the atmosphere. Shallow groundwater (water less than 20 feet [6 m] below the surface) flows downgradient through the pore spaces in the soils until it surfaces in a stream, wetland, or swale, sometimes in the form of a seep or spring.

Precipitation falling onto naturally impervious surfaces (bedrock), however, flows along the surface. Precipitation also flows along the surface if the soils become saturated and cannot hold any more water.

3.2.3.2 Movement of Water in Disturbed Landscapes

Human activities on the land change the movement of water across and through the landscape such that there are significant changes in runoff patterns and hydroperiods in a watershed (Booth 1991, Vought et al. 1995, Azous and Horner 2001). Surface runoff, rather than infiltration, comes to dominate water flows, as shown conceptually in Figure 3-2. Infiltration and subsurface flow can be altered by any of the following conditions:

- Removal of vegetation
- Compaction of soil (through grazing, earthwork, lawns, or playfields)
- Reduction in size of soil particles or the spaces between particles (through tilling or grading)
- Reduction in the organisms that aerate the soil
- Placement of drain tiles, ditching, road cuts, utility lines
- Construction of impervious surfaces

These conditions result from human land uses such as agricultural conversion, urbanization, and forest practices (Dunne and Leopold 1978, Booth 1991, Euliss and Mushet 1996). The disturbances to water movement and sources created by specific land uses are described later in this chapter, whereas literature on the resulting impacts on wetland functions is synthesized in Chapter 4.

Removing vegetation allows precipitation to reach the soil surface faster, and therefore soil saturation occurs more rapidly. As soils become saturated, additional precipitation accumulates more rapidly on the surface and moves as sheet or surface flows. When soils are compacted, the precipitation cannot enter the soils readily and surface water accumulates more rapidly. Loss of permeability in the soil can persist even after compacted soils become vegetated as urban lawns, playfields, and in some agricultural conditions (Dunne and Leopold 1978).

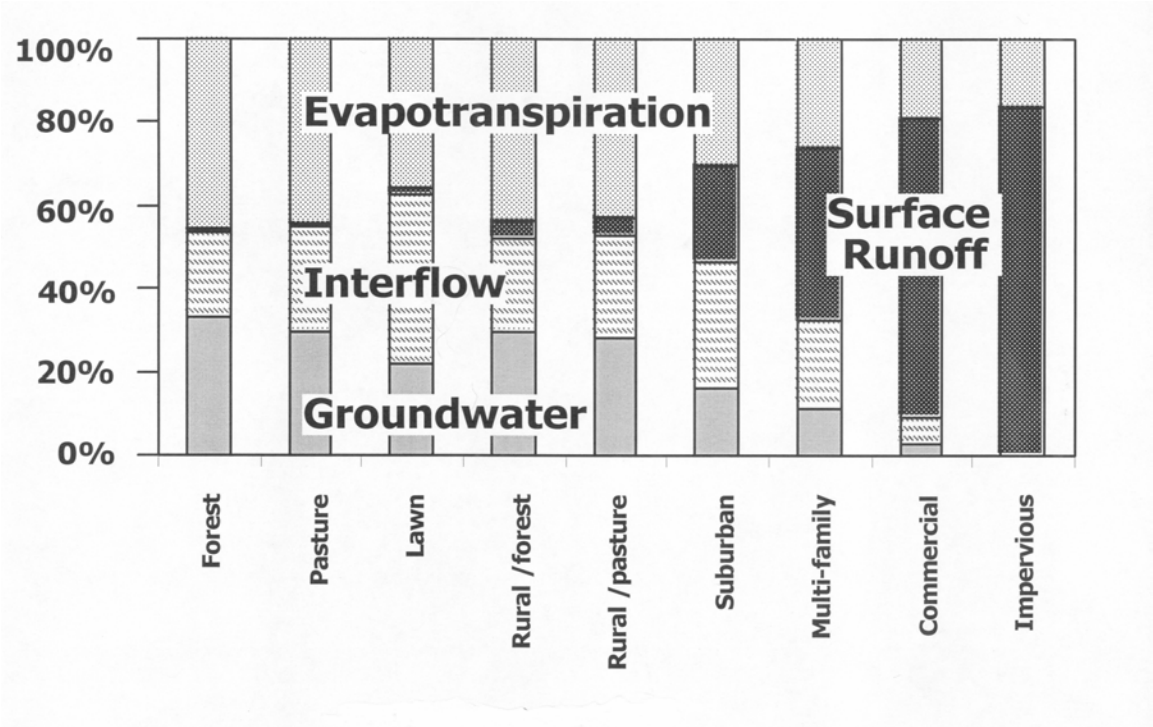

Figure 3-2. Changes in the proportion of groundwater, interflow, evapotranspiration, and surface runoff with different types of land cover. Source: Beyerlein (1999). Reprinted with permission.

Under any of these conditions, runoff essentially becomes surface flow. Scientists call this "saturated overland" or "Hortonian" flow. Water flowing along the surface carries sediment and any other dissolved or adsorbed materials downgradient more rapidly than if the water is allowed to infiltrate in undisturbed soils (Ziemer 1998).

In general, alteration of water flow results in a "shortening" of the path that water would naturally follow on its route through a watershed. It reduces the residence time of water in the ground and in bodies of surface water, such as streams or wetlands, within the watershed.

Changing the water flow also results in increased rates and volumes of stormwater and changes the timing of stormwater entering aquatic systems. This can have numerous effects on aquatic systems as described in Section 3.4.2 on the effects of urbanization. For example, these changes circumvent or reduce:

- The removal of nutrients, pathogens, and toxics in the soil
- The filtering of sediment from surface flows through vegetated buffers and wetlands
- The reduction of downstream peak flows

3.2.3.3 Thresholds at Which Impervious Surfaces Significantly Impact Aquatic Resources

According to research throughout the country and in the Pacific Northwest described below, the degree of alteration in hydrologic processes and the subsequent impacts to aquatic habitats (including wetlands) is governed by the percent of impervious area and the percent of forested cover within a watershed. When soil is covered with impervious surfaces there is no opportunity for infiltration. All precipitation that falls on an impervious surface becomes surface water which flows downgradient.

Research in western Washington generally indicates that 10% or more impervious cover within a watershed can result in significant impacts to the habitat structure and function of freshwater aquatic systems (Azous and Horner 2001). Schueler and Holland (2000) identified a threshold of less than 10% basin imperviousness before stream channel degradation occurred. Reinelt and Taylor (2001) discovered that 20% impervious cover from upstream development increased the peak and volume of stormwater runoff to the point that it began to dominate the hydroperiod of downstream wetlands.

The term "impervious surface" as used in the literature here means more than just a hard impermeable surface such as an asphalt parking lot. There are many actions humans take that reduce the permeability of soils, and these are included in the calculations of "percent impermeable surface." For example, compacted soils found in lawns and landscaped areas function just as impervious surfaces during storm events (May 1996). May (1996) also cites a study by Wigmosta and Burgess (1990) that found that lawns contributed up to 60% of the runoff in a sub-basin during a storm event.

Total impervious area (TIA) is sometimes challenging to assess without high-resolution aerial photographs and accurate GIS mapping capabilities (especially in watersheds with extensive forested coverage). Reinartz and Warne (1993) found that using road density as an indicator of basin impervious area resulted in findings nearly identical to those resulting from estimation of imperviousness from aerial photographs.

Reinelt and Taylor (2001) concluded that removing as little as 3.5% of the forested cover in a rural, low-density residential area resulted in changes in the pattern of water movement in the basin. Looking at percent forested cover in the Puget Sound Basin, Booth et al. (2002) have determined that natural hydrologic processes are maintained if 65% of a watershed remains in a forested condition.

Because each watershed has different physical, chemical, and biological characteristics and patterns of impervious cover, the threshold at which aquatic resources experience significant effects will vary. There is no research, for example, on the accuracy of the 10% threshold for non-forested watersheds in eastern Washington.

Table 3-2 summarizes additional findings on the effects of impervious cover on various biological characteristics of watersheds. As noted previously, specific impacts to wetland functions are described in Chapter 4.

Table 3-2. Summary of findings on the impacts of impervious cover.

Reference	Impacts to:	Key Finding
Booth (1991)	Fish habitat; channel stability	Channel stability and fish habitat quality declined rapidly with over 10% impervious cover
Luchetti and Fuerstenburg (1993)	Fish populations	Shift from coho to more tolerant cutthroat trout at 10 to 15% impervious cover
Taylor (1993)	Wetland plants and amphibians	Mean annual water level fluctuations are inversely correlated to density of plants and amphibians. Sharp declines occur when impervious cover exceeds 10%
Limburg and Schmidt (1990)	Fish spawning	Resident and anadromous fish eggs and larvae declined in 16 tributary streams with over 10% impervious cover
Shaver et al. (1994)	Invertebrates	Insect diversity dropped sharply at 19 streams with 8% to 15% impervious cover
Steedman (1988)	Invertebrates	Negative correlation between biologic integrity and increasing development at 209 streams. Degradation started at 10% impervious cover

3.2.4 General Disturbances to the Quality of Water at the Landscape Scale

Two principal mechanisms have been documented to describe how land uses in a watershed change water quality in that watershed:

- Changes in hydroperiod increase erosion and sedimentation (Booth 1991, Booth and Reinelt 1993, Horner et al. 1996).
- Human uses of the land generate pollutants that are then transported into aquatic systems (Reinelt and Horner 1995).

Larger volumes of water, moving at faster rates, scour channels and cause rills in unvegetated soils. Moving water picks up and transports sediment and the pollutants associated with sediment particles. In addition, research shows that water flowing across the ground surface tends to pick up and convey dissolved nutrients and toxics directly into receiving waters (Young et al. 1980, Emmett et al. 1994, Gilliam 1994, Brenner 1995, Reinelt and Horner 1995, Vought et al. 1995, Crosbie and Chow 1999, Sheridan et al. 1999, Azous and Horner 2001).

Pollution conveyed by surface runoff (called non-point-source pollution) has been identified as the dominant source of pollutants in surface water. Non-point pollution is not discharged from the "end of a pipe" such as a large factory. Instead it is caused by sediment, metals, excess nutrients, and bacteria from a variety of dispersed sources (Reinelt and Horner 1995) such as stormwater, contaminated runoff from urban settings, agricultural runoff, and construction runoff (Baker 1992). These pollutants can have numerous impacts on wetlands and their functions as described in Chapter 4.

3.2.5 General Disturbances to the Connectivity of Habitats at the Landscape Scale

Human activities within a watershed often break up ecosystems into small patches that are separated by roads, buildings, or tilled fields. This break in the historic connections within and between ecosystems is a disturbance that is called fragmentation. "Fragmentation is the piecemeal disassembly of terrestrial habitats into discontinuous, oftentimes isolated, patches as a consequence of human development" (Towson University no date). Habitat fragmentation also stems from habitat loss as land that once provided habitat for wildlife is converted to other uses (Heinz Center for Science 2002).

Suburban and urban development, farmlands, roads, railroads, powerline corridors, and other land uses cause various kinds and degrees of fragmentation (Heinz Center for Science 2002), and these are discussed in more detail in subsequent sections of this chapter. In addition, human activity can create landscapes that are less varied than the landscapes historically present in a watershed. Particularly in the West, natural fires create a patchy landscape, where forest and grasslands are intermingled in a mosaic. Fire suppression and the large fires that result after long periods of suppression can create broad expanses of very similar vegetation (Heinz Center for Science 2002).

Fragmentation can reduce habitat for wildlife species that require larger, connected patches. It can hinder the movement of some species and introduce predators, parasites, and competitors associated with different land uses. The fragmentation caused by the major land uses is described in this chapter, and the impacts of fragmentation on the functions of wetlands are described in Chapter 4.

3.2.6 Summary of Key Points

- Many human land uses change the movement and sources of water in a watershed, thereby creating a disturbance that affects the performance of functions in wetlands.
- In general, alteration of water flow by human uses of the land results in a "shortening" of the path that water would follow on its route through a watershed. It reduces the residence time of water in the ground and in the bodies of surface water, such as streams or wetlands, within the watershed.

- Changes in the amount of water and the frequency and fluctuations in water volumes can also change how sediments, nutrients, and toxic contaminants come into a wetland.
- When the amount of impervious surface in a watershed reaches a threshold, significant effects on aquatic resources begin to appear. While 10% or more impervious cover is generally cited as a threshold, the differing conditions in individual watersheds and regions mean that the threshold for impacts will vary.
- Two principal mechanisms have been documented to describe how land uses in a watershed change the water quality in that watershed: (1) land uses increase erosion and sedimentation, and (2) human uses of the land generate pollutants that are then transported into aquatic systems.
- Human activities within a watershed often break up ecosystems into small patches and this disturbance is called fragmentation.

3.3 Disturbances Caused by Agriculture

In western Washington, sites with flat topography suitable for agriculture were often located in river or stream floodplains. Many areas of these floodplains were wetlands with high water tables that persisted late into the growing season. As noted by Bardecki (1988) in a study in southern Ontario, the human response to the presence of excessive soil water was usually to drain the soil. In addition to draining wetlands, agricultural practices in some parts of the state may have resulted in the creation of wetlands or expansion of pre-existing wetlands through the introduction of irrigation water runoff.

This section describes the types and severity of disturbances that can be caused by agricultural practices. As mentioned previously, these disturbances can, in turn, affect factors that control wetland functions.

Agricultural practices play a significant role in influencing water movement in many regions of Washington. However, much of the research on wetlands in Washington over the last 10 years has been on the effects of urbanization. Although some of the consequences and effects of agriculture and urbanization may be the same or similar, they are presented separately here to clarify what the literature states for each type of land use.

3.3.1 Changes in the Physical Structure within Wetlands Resulting from Agricultural Practices

Agriculture disturbs wetlands directly through conversion of the wetland to fields or pasture. For example, tilling the soil within a wetland will disturb its structure (Nowak 1980, Hayes 1995). Other conversion activities include filling or tilling, draining through

tiles or channels, or removing the wetland vegetation and planting upland vegetation. Livestock grazing in streams and riparian wetlands also has well documented effects on the structure of plants and soils in wetlands as described below.

In studying riparian wetlands, Chappell et al. (2001) concluded that wetland loss in western Washington has been caused primarily by conversion to land development and agriculture. Although Chappell et al. (2001) do not estimate the loss that can be attributed to the different types of land uses, Bell (2002) found that 40% of the losses to peat wetlands in King County between 1958 and 2000 could be attributed to agriculture.

Eastern Washington riparian wetlands have been reduced from an estimated 2% cover of the landscape to 0.5%, with most of the remaining riparian wetlands at high elevations (Chappell et al. 2001). While new wetlands have developed in some areas of the Columbia Basin due to irrigation, there is still a net loss of wetland acreage in eastern Washington mainly because of road building, agricultural conversion, and flood control (Chappell et al. 2001). The studies of wetland losses in Washington do not further distinguish how much wetland has been converted to various land uses. It can be assumed, however, that agriculture has resulted in significant losses because this is a continental trend as summarized below.

Outside of Washington, tremendous loss of wetland acreage has been attributed to agricultural filling, draining, and ditching in the prairie pothole regions of North America (Leitch 1983, Tiner 1984, Turner et al. 1987, Bardecki 1988). Researchers in Canada estimated 73 to 95% of the original wetlands in the area studied had been lost to agricultural conversion by the late 1960s (Snell as quoted in Bardecki 1988). Their work in Canada parallels the findings of Tiner (1984) that up to 87% of wetland loss in the United States was related to agricultural practices.

Bardecki (1988) found the greatest extent of wetland loss had occurred in areas with agricultural drainage. He also noted there was no limit to wetland loss. Conversion for agricultural use would occur until all lands suitable for crop production were in use. Of significance is his observation that drain tiles, drainage ditches, and stream channelization often resulted in far greater area of wetland impact than documented by various studies. Many studies only recorded the loss of wetland acreage if it was directly converted to agricultural uses but they failed to identify ancillary hydrologic impacts. He cited the example of construction of channels and ditches through existing wetlands downgradient of the original fields slated for drainage. The presence of the drainage channels often caused undocumented adverse effects in the remaining wetlands.

The literature on the effects of grazing is focused primarily on riparian habitats, including riparian wetlands. Only a few of the studies found on this topic are located in the Pacific Northwest, but much of the literature from the Midwest and even some from Australia may also be relevant because the types of disturbances caused by grazing are not geographically isolated. Many of the studies focused on riparian areas without differentiating between upland and wetland areas, and many focused on streams.

In summary, the effects of grazing in riparian areas include (Armour et al.1991, Busby 1979):

- Loss of vegetation
- Trampling and related sloughing and erosion of streambanks
- Shallower and wider streams
- Elevated water temperature
- Sedimentation

From a study in central Canada's prairie pothole region, Turner et al. (1987) determined that activities such as haying and grazing may cumulatively have the same ultimate consequence as filling or draining.

The effects of grazing on riparian vegetation vary significantly depending on the frequency and intensity of grazing (Clary 1995, Clary et al. 1996, Jansen and Robertson 2001). Soil compaction and a reduction in ground-cover vegetation lead to erosion and greater volumes of runoff from the compacted areas. Also, as native plant species are trampled and grazed and shading is reduced, there is more opportunity for establishment of species that can tolerate disturbance (see Chapter 4).

Cranberry growing operations in Washington are another type of agricultural land use that affects wetlands. However, cranberry production is limited to very small areas along the southern Washington coast in Pacific and Grays Harbor Counties. The types of impacts that occur from conversion of wetlands to cranberry production are very different from other types of agricultural impacts. Due to the limited area affected, and time and funding limitations, this report does not attempt to address the effects of cranberry production on wetlands in the state.

3.3.2 Increased Amounts of Water in Wetlands Resulting from Agricultural Practices

Water availability was a limiting factor for agricultural practices in the areas of low rainfall until the U.S. Bureau of Reclamation began intensive damming and irrigation projects in the early 1900s (Lemly 1994). Since then, irrigation practices have been influencing wetland presence and functions in areas in the rain shadow of the Olympic Mountains and the arid parts of eastern Washington. Most of the scientific literature concerns western states such as Colorado and Wyoming, as well as Washington east of the Cascade Mountains. No information was found regarding the disturbances caused by irrigation practices on the Olympic Peninsula.

Irrigation can increase the amount of water at or near the surface (Adamus 1993). This may result in the creation and maintenance of wetlands in locations where they did not previously exist. New wetland areas have formed because of the sustained higher water table from seepage out of irrigation reservoirs, irrigation channels, and irrigation runoff. Leakage from irrigation channels and ditches often allows the formation of wetlands along channel margins or immediately downslope of ditches. Excess irrigation water

applied to fields that exceeds the capacity of the soils to absorb water ("tailwater") may also form wetlands in low-lying areas that collect excess runoff. Tailwater also includes the spillage that occurs during operation of the irrigation system (Adamus 1993). For example, the Potholes Reservoir area within the Columbia Basin Irrigation Project contains wetland complexes that exist because of high groundwater caused by the high water levels in the reservoir (Tabor, personal communication).

Studies in Wyoming by Peck and Lovvorn (2001) support the idea that irrigation can be significant in creating and supporting wetlands and the biotic communities that depend upon them. The authors noted that 65% of inflows into wetlands in the Laramie Basin were derived from irrigation waters. They reached this conclusion by studying the loss of wetlands when irrigation practices were made more efficient (this is discussed further in Section 3.3.3).

In some instances, pre-existing wetlands experience deeper water for longer durations in the summer due to runoff from irrigation. Wetlands in the Potholes Reservoir that may have been seasonally inundated have become permanently inundated because of irrigation (Creighton et al. 1997).

In Colorado, Adamus (1993) differentiated between types of irrigation-related wetlands during a study of bird use of wetlands associated with irrigation waters. His work is cited here for relevant insights into the complexities of wetlands associated with irrigation. However, due to physiographic and climatic differences between the Colorado Plateau and Washington, not all of his findings may be directly relevant to irrigated agricultural lands of the state. He identified the following types of irrigation-related wetlands:

- **Irrigated wetlands** are those that are created on farmed lands as the result of the duration and frequency of inundation from irrigation waters. The wetlands are most often created on the farmed lands within the actual zone of irrigation.
- **Enhanced wetlands** are those that are enlarged or their hydrologic regime extended (longer inundation or saturation) as the result of runoff from irrigation waters.
- **Induced wetlands** are those that develop as a result of irrigation runoff (from the farmed lands) where wetland did not exist previously. These wetlands may or may not be located on the lands that are irrigated, but the source of the runoff water that creates these wetlands is excess runoff from irrigated fields.

Adamus (1993) also noted:

> *However, even after visiting a site it is difficult to determine conclusively the primary source of water that sustains a wetland. Irrigated wetlands, as considered by this project, can range from wetlands that are completely supported by irrigation runoff at all seasons, to wetlands that exist naturally but for which any measurable amount of their water originates from irrigation, however indirectly (e.g., through seepage or raised water tables). In this sense, virtually all wetlands in irrigated regions could be*

> *considered 'irrigated wetlands.' …determining whether the primary water source of a wetlands is irrigation-related in many cases requires considerable judgment, and no highly replicable approach exists that is applicable to all situations.*

Adamus (1993) determined that the following are *not* adequate criteria to distinguish the water source and, therefore, the origin of a wetland:

- **Seed species richness:** Wetlands that are the result of irrigation water and are more than a few decades old are difficult to distinguish from pre-existing wetlands based on the species richness of the seed bank.
- **Organic content of soils:** Organic material is not an appropriate indicator of water origin. Organic detritus likely accumulates at different rates based on a variety of influencing factors. Much of the organic detritus appears to mineralize by the end of the growing season.
- **Presence of large willows and black cottonwoods:** A lack of large mature stands of black cottonwood and willows is also not an indicator of pre-existing vs. irrigated systems. Cottonwood stands may have been harvested or may never have become established. Anecdotal information concludes that cottonwood regeneration may not occur as frequently in irrigated wetlands due to overgrazing and the effects of flood management.

In the Wyoming setting studied by Peck and Lovvorn (2001), salinity of groundwater was also a factor in wetlands receiving shallow groundwater inputs from irrigated fields. Vegetation and biotic communities in the wetlands were correlated to both the water availability and the relative salinity of waters reaching wetlands. In summary, "…different irrigation practices have contrasting effects on a range of wetland types. These effects will change seasonally to impact different organisms with varying life histories, flooding requirements, and salinity tolerances" (Peck and Lovvorn 2001). The effects of changes in salinity are discussed in Section 3.3.8.

3.3.3 Decreased Amounts of Water in Wetlands Resulting from Agricultural Practices

Creighton et al. (1997) note that extensive areas of the landscape in the Columbia River Basin of eastern Washington have been altered by irrigation. One result of irrigation projects, they note, was "a sharp reduction in the amount of water available to native wetlands." In some instances sources of fresh water for wetlands, not resulting from irrigation, were diverted for agricultural uses and less water reached the wetlands.

In Wyoming, Peck and Lovvorn (2001) investigated the potential consequences of increasing efficiency in irrigation practices by lining ditches and using sprinkler systems (rather than flooding the fields). The authors noted that 65% of inflows into wetlands in the Laramie Basin were derived from irrigation waters. Therefore, with increased efficiency of water used for irrigation, the presence of wetlands in irrigated arid lands

could decline. (The Wyoming data may be relevant to eastern Washington although the underlying geology and irrigation practices may not be identical.)

In California, the drought of 1985 through 1992 resulted in implementation of greater water conservation measures and therefore a decrease in the production of irrigation tailwater. There was a subsequent decrease in the volume of water reaching wetlands. (Creighton et al. 1997).

Lower water levels in a wetland can also result from the direct ditching and draining for agricultural purposes. In this case the water entering the wetland is not reduced, rather it is shunted through the wetland and the storage capacity of the wetland is diminished. The ditching may be so effective that the area becomes an upland. If, however, the draining is only partial the wetland may remain, but with lower water levels and probably a reduced area. The literature review did not disclose any information on how many wetlands in Washington may be impacted in this way.

3.3.4 Increased Fluctuations of Water Levels in Wetlands Resulting from Agriculture

The findings of Euliss and Mushet (1999) in North Dakota on the effects of agriculture on water level fluctuations in wetlands are probably significant for wetlands in the arid grasslands of the Columbia Basin. These areas have similarities in precipitation and geologic patterns. These authors found that the hydroperiods for temporary, seasonal, and semi-permanent wetlands were all significantly affected by agricultural practices within the wetland's contributing basin. There was a three-fold increase in water level fluctuations of wetlands within tilled agricultural landscapes (average 5.5 inches [14 cm] fluctuation) compared to those surrounded by natural grasslands (average 1.6 inches [4 cm] fluctuation). The authors concluded, "Tillage reduces the natural capacity of catchments to mitigate surface flow into wetland basins during precipitation events, resulting in greater water level fluctuations in wetlands with tilled catchments."

3.3.5 Increased Input of Sediment Resulting from Agriculture

Tillage and grazing adjacent to a wetland or in a watershed can disrupt the soil, creating a source of sediment for surface runoff to transport downstream into wetlands and other aquatic systems. In addition, ditching wetlands in agricultural areas increases the rate of water movement by removing or reducing vegetation that acts to decrease the velocity of water. Unvegetated channels and ditches may be the source of sediment through increased erosion within the ditch (Brown 1988).

Baker (1992) compared sediments in agricultural runoff to those of wastewater plant effluent. He found that agricultural runoff can have suspended solids in the range of 100 to 1,000 milligrams per liter (mg/l), compared to less than 30 mg/l for wastewater that had received secondary treatment. Baker (1992) also found that non-point-source

pollution from agricultural lands is driven by storms. It is therefore highly variable in extent and timing. He noted that in agricultural settings large storms can increase the sediment load by two to three orders of magnitude in a year, while the loads in wastewater discharge remain relatively consistent.

Wind-borne sediments that are eroded from tilled fields also generate high sediment loads to wetlands and streams in eastern Washington. The U.S. Department of Agriculture estimates that about half of the total farmland in Washington lost more than 2 tons of soil per acre per year through the action of wind in 1997. About 10% of the total farmland lost more than 10 tons per acre per year (Natural Resources Conservation Service 1997). By adding up the estimates of erosion rates and area that is farmland, it can be estimated that in 1997 about 15 million tons of topsoil were lost through wind erosion from fields in the state.

Sediment will eventually be transported into rivers and streams or deposited in wetlands. Wetlands found in depressions are often the low points in a landscape and will receive sediments that fall in the surrounding areas. The field teams that are calibrating both the methods for assessing wetland functions and the Washington State Wetland Rating System have observed wind-blown sediments in many wetlands of eastern Washington that were several inches deep.

3.3.6 Increased Input of Nutrients Resulting from Agriculture

In the United States the export of phosphorus and nitrogen from agricultural land can be three times higher for phosphorus and 12 times higher for nitrogen than from forested lands (Omernik 1977). Many of these nutrients are transported to wetlands, streams, rivers, and lakes because they are washed out of fields or infiltrate into groundwater. In Washington State, Williamson et al. (1998) found elevated levels of nutrients in the groundwater below irrigated fields on the Columbia Plateau. Their assumption is that the source of these nutrients is their application to fields above the groundwater.

The changes in the input of nutrients as a result of agriculture are illustrated by a study in Estonia in eastern Europe that documented what happened when agricultural fertilizers were no longer placed on agricultural lands. There was a four-fold to 20-fold decrease in pollutants associated with agricultural runoff after the collapse of agricultural collectives and the subsequent decline in the application of commercial fertilizers and manure (Mander et al. 2000). Based on 10 years of data (1987 through 1997), the researchers determined that total nitrogen, total phosphorus, biochemical oxygen demand (BOD), and sulfate all declined significantly with the demise of agricultural practices in the contributing watersheds. Forested portions of the watersheds upstream of the agricultural lands did not experience measurable changes in water quality parameters, eliminating the possibility that climatic change was the cause.

3.3.7 Increased Input of Toxic Contaminants Resulting from Agriculture

Several authors have identified agriculture across the country as one of the primary causes of non-point-source pollution in aquatic systems (Brenner 1995, Reinelt and Horner 1995, Thurston 1999). Agricultural chemicals are used to control noxious weeds, insect pests, and damaging fungi and bacteria.

Agricultural chemicals applied to fields enter downstream aquatic resources such as wetlands through three primary pathways (Neely and Baker 1985):

- Adsorbed to sediment particles
- Dissolved or suspended within surface flows
- Dissolved within subsurface drainage

Farming practices and the type of chemicals used determine how the pollutant is transported into wetlands. For example, some herbicides applied to corn are water soluble. Neely and Baker (1985) reported that water flowing across crop residue left after harvesting may wash off remnant herbicides. The concentration of such an herbicide in wetlands downgradient of a corn field may increase as a result. Similarly, Donald et al. (1999) documented that wetlands in the Canadian prairie pothole region receive high levels of pesticides when pesticides are applied to fields prior to significant rains (precipitation totaling more than 2 inches [50 mm] after application). Another study, in California's Central Valley, found that surface water runoff from irrigated fields could have elevated levels of pesticides and herbicides if there had been aerial application of the chemicals or a recent land-based application (Lemly 1994).

Subsurface drainage may also contain pollutants at low levels. Lemly (1994) reported that subsurface waters from irrigated fields had low levels of herbicides or pesticides. These substances were removed from the water column through adsorption as the water filtered through the soils before draining into the subsurface collection system. Williamson et al. (1998) found elevated levels of pesticides in the groundwater below irrigated fields on the Columbia Plateau.

3.3.8 Increased Levels of Salt Resulting from Agriculture

Agricultural practices in irrigated areas can increase the salt content of water in a watershed or in areas immediately adjacent to a field. This means that wetlands receiving water from irrigated areas may also be subject to higher salt concentrations.

The soils in dry areas have developed in an environment of limited rainfall and significant periods of drying. In these areas the rate of evapotranspiration is higher than rainfall, and this draws water from below the ground's surface and causes many soluble minerals to accumulate in the upper soil horizons (Caltech 2003).

Soluble salts in irrigation water will be deposited in soils near the root zones of plants because much of the water in arid regions is lost by evaporation rather than downward transport. This salinization occurs with nearly any type of irrigation. Even if the irrigation water is only slightly saline, repeated cycles of evaporation lead to build-up of toxic salt levels in the soil (Caltech 2003). Irrigation return waters are often high in salt content (Adamus et al. 2001).

3.3.9 Reduced Connectivity (Fragmentation of Habitat) Resulting from Agriculture

No information specific to the disruption of the connections between wetlands and other habitats (fragmentation) resulting from agricultural activities was found in the literature. It can be hypothesized, however, that such fragmentation has occurred. The direct loss of wetlands through conversion to other uses such as agriculture increases fragmentation by removing natural cover within the wetland itself. The conversion of wetlands to agricultural uses is discussed in Section 3.3. In addition, most agricultural practices remove existing "natural" vegetation that acts as a corridor between and through wetlands. This in turn can have an impact on the movement of plants and animals (as described in Chapter 4).

3.3.10 Other Disturbances Resulting from Agriculture

Several other types of disturbances that have been attributed to agricultural activities:

- Alteration of soils
- Construction of roads
- Noise
- Invasion by exotic plant and animal species

These disturbances are not discussed in detail in this chapter because little information was found describing how agricultural practices create these disturbances. The impacts of these disturbances, however, have been documented and are summarized in Chapter 4.

3.3.11 Summary of Key Points

- Agriculture may affect wetlands directly through conversion of the wetland to fields or pasture. This is often done by direct filling or tilling, by draining through tiles or channels, or by removing the wetland vegetation and planting upland vegetation.
- Livestock grazing in streams and riparian wetlands also has documented effects on the physical structure of wetlands.

- Irrigation can result in the creation and maintenance of wetlands in locations where they did not previously exist. This is a controversial regulatory issue in areas of the state that are irrigated.
- Conversely, agriculture can reduce the amount of water available to wetlands by either diverting water that would otherwise reach pre-existing wetlands, or imposing more efficient irrigation practices that reduce the amount of leakage reaching irrigation-dependent wetlands.
- Wetlands in tilled areas may experience greater water level fluctuations.
- Disruption of the soil through tilling and grazing can create a source of sediment than can be transported further downgradient. Sediments may also be carried by winds from tilled fields.
- Agricultural areas can have an increased load of nutrients and pesticides in surface runoff and groundwater.
- Agricultural practices in irrigated areas can lead to accumulation of salts in the upper soil horizons. Irrigation may leach out the accumulated salts.
- Fragmentation of wildlife habitat is a secondary consequence of loss of wetlands through agricultural practices. Clearing land for farming removes natural cover and connections between habitats.

3.4 Disturbances Caused by Urbanization

Urbanization causes disturbances that affect the controls of wetland functions, and thereby the performance of those functions, both at the scale of the watershed and within individual wetlands. These disturbances impose a variety of changes that profoundly affect watershed processes and, therefore, the downgradient drainage system and the wetlands found there. Changes include clearing of vegetation, compaction of soil, modifications to water conveyance, alterations to riparian corridors, human intrusions, introduction of chemical contaminants, and increased areas of impervious surface. A summary report by the U.S. Environmental Protection Agency (1993) concludes that urbanization strongly affects water movement within a watershed by increasing rates of surface flow, reducing subsurface volumes, and reducing baseflow. These pervasive, landscape-level changes commonly affect virtually all areas of an urban watershed (Dunne and Leopold 1978, Booth 1991, Booth and Reinelt 1993, Hollis and Thompson 1998).

Much of the scientific research on urbanization in the Pacific Northwest comes from the Puget Sound Wetlands and Stormwater Management Research Program initiated in 1986 in King County. Published results include theses by Azous (1991), Chin (1996), Ludwa (1994), May (1996), and Taylor (1993). The book *Wetlands and Urbanization: Implications for the Future,* edited by Azous and Horner (2001), is a summary of the significant findings of the research.

3.4.1 Disturbances to the Physical Structure of Wetlands Resulting from Urbanization

Approximately 13% of the wetland losses in the United States can be attributed to urbanization, road building, and other types of conversion (Tiner 1984). Kusler and Niering (1998) estimate that 85% of the wetlands in urban areas of the nation have been destroyed, and most of the remaining 15% are moderately to severely impaired in function. Data specific to Washington are very limited. One study (Bell 2002) found urban and residential development in King County accounted for 28% of the peat wetlands lost between 1958 and 2000.

This trend will likely continue. It is estimated that more than 80% of the U.S. population will be living in urban areas by 2025, up from 74% in 1989 (Gerguson and Robinette 2001). Increases in urban population are generally accompanied by increased development density and sprawl. Wetlands in these areas are either converted to urban land uses or, if they are not directly disturbed, are degraded through a variety of causes as described in the following sections.

3.4.2 Increased Amount of Water in Wetlands Resulting from Urbanization

Urbanization is recognized as both increasing and decreasing the flows that reach downgradient aquatic systems such as wetlands. Greater volumes of water are generated more quickly while smaller, long-duration flows that would occur under less developed conditions are reduced or perhaps eliminated. Research has shown that collecting stormwater through modern storm drains, culverts, and catchments results in the rapid transport of large volumes of stormwater runoff into rivers, lakes, and wetlands at much faster rates and higher volumes than under predevelopment conditions (Dunne and Leopold 1978, Booth 1991, May 1996). Although some of the research has focused on the effects of urbanization on streams, the findings on changes in flow volumes, rates, and frequency apply equally to wetlands that receive storm drainage. Streams and wetlands are "intimately interconnected in the watersheds of western Washington" (Booth 1993).

Research conducted in the Puget Sound lowlands has shown strong correlations between the effects of urbanization in a watershed and the hydrologic regime in that watershed.

The amount of impervious surface within a contributing basin is a key influence on hydrologic patterns, and even small changes in watershed conditions have measurable influences on the flows and volumes of water in the system. (See Section 3.2.3.3 for further discussion on thresholds at which impervious surface affects aquatic systems.)

3.4.2.1 Increased Frequency of Erosive Flows

One consequence of urbanization is an increase in the frequency of erosive flows within a watershed. As reported by Booth (1991), several studies concluded the most common effect of urbanization was an up to five-fold increase in peak flow rates from a given storm event. The largest relative increases in erosive flows were found for the smallest storm events. This is very significant because small storm events are the most frequently occurring storms. A small storm event is the "two-year-event": a storm with a given volume of rain falling within a 24-hour period that has the statistical likelihood of occurring every two years (the statistics are based on over 40 years of measured rainfall). That means that small storm events have the greatest percent increase in flows over natural conditions, and frequent small storms have the greatest relative increase in erosive flows. Contrary to what might be expected, it is these recurring small storms that have the greatest cumulative effect on erosion and sedimentation, not the large, less frequent storm events (Booth 1991).

Thus, larger volumes of water enter channels and wetlands more rapidly after a given storm event in a basin where the removal of forests and the increase in impervious surfaces have altered hydrologic processes (Booth 1991). After an area has been developed and the forest canopy removed, high rates of flow continue for a longer duration. These flows may carry sediment and other pollutants into downgradient wetlands.

3.4.2.2 Increased Volume of Runoff and Longer Duration of Flows

Booth (1993) notes that a basin with increased imperviousness will experience an increase in the magnitude of runoff volume from a given storm event. The "typical" event occurs far more frequently. For example, the peak flows created from a two-year storm event, after urbanization, will occur far more frequently than every two years. Small storm events that did not create measurable peak discharges in forested conditions create measurable peak runoff flows in urbanized conditions, because the removal of the forest canopy makes the same size storm event result in far greater volumes of water reaching aquatic resources such as wetlands and streams. Modeling based on detailed data from basin monitoring identified that larger flows with more erosive force may occur in urbanized basins with much greater frequency, for example increasing from once or twice per decade to several times per year.

In urbanizing watersheds, stormwater ponds are designed to hold the excess *volume* of stormwater generated from the impervious surfaces. The ponds are designed to release stormwater at the same *rate* as that modeled for the natural vegetated basin for a given

storm in pre-existing conditions (Booth 1991). However, in order for the ponds to discharge the increased volume of water at the same low rates, they must take more time, or cause an increased *duration* of flows.

3.4.2.3 Consequences of Changes in Water Regime

The consequences of the interplay between rates, volumes, and durations are complex. Research on the impacts of urbanization on stormwater and watershed processes indicates that urbanization results in several disturbances that can impact wetlands (Booth 1991, Azous and Horner 2001, Reinelt and Taylor 2001, Thom et al. 2001):

- Increased erosion
- Sediment movement and deposition
- Burying of vegetation
- Increased depths of inundation
- Water level fluctuations
- Downcutting of natural channels (which can remove riparian vegetation from the floodplain)
- Changes in the seasonal extent and duration of saturation and inundation
- Unstable substrates

Urbanization can also cause a decrease in interflow (shallow subsurface flow) and base flow from the urbanized watershed (U.S. Environmental Protection Agency 1993). Changes in the volume of interflow may influence the hydroperiod of downgradient wetlands if they are fed by that shallow subsurface flow.

Roads and parking lots are an important component of the impervious surface area in a watershed. The City of Olympia in 1994 determined that transportation features (roads and parking lots) typically composed between 63 and 70% of total impervious area within suburban areas (Schueler and Holland 2000).

Although changes to the water regime have been documented as a general impact of urbanization, little information is available on how it might affect wetlands. In one study, "drier-end" wetlands were more frequently destroyed on the edge of the urban fringe of the City of Portland (Holland et al. 1995). The study attributes 27% of the total losses to "drought," while urbanization caused the loss of 63% of the total. However, it remains unclear in the study whether these wetlands were actually lost or perhaps not recognizable in drought conditions.

3.4.3 Increased Fluctuations of Water Levels Resulting from Urbanization

Reinelt and Taylor (2001) used water level fluctuation as the primary measure of wetland hydroperiod, stating: "Water level fluctuation is perhaps the best single indicator of wetland hydrology, because it integrates nearly all hydrologic factors." They documented four factors in a wetland and its watershed that have the strongest influence on water level fluctuations:

- Forest cover in the watershed
- Impervious cover in the watershed
- Constriction of the wetland outlet
- Ratio of wetland to watershed area

Wetlands in basins with the highest degree of impervious area had the highest water level fluctuations. Wetlands in basins with 90% or more forested land cover and less than 3% impervious area generally exhibited smaller ranges in water level fluctuations (Reinelt and Taylor 2001). Further information on thresholds at which impervious surface influences aquatic ecosystems is provided in Section 3.2.3.3.

Wetland size is also important in determining the effects of urbanization on water level fluctuations. Reinelt and Taylor (2001) observed that wetlands that were small in relation to their contributing watersheds had greater water level fluctuations and were dominated by surface inflow. Wetlands that were larger in comparison to their contributing watersheds had smaller water level fluctuations and more groundwater influence. Wetlands with a constricted outlet (undersized culvert, beaver dam, or embankment) had a greater water level fluctuation than wetlands with less constricted outlets.

Stormwater runoff from urbanization, as well as other land use alterations, frequently causes several changes in how water levels fluctuate in wetlands. All aspects of fluctuations in water levels are changed by urbanization:

- The **magnitude** of the effect of storms is changed by causing a two-year event to act like a larger storm. A larger volume of water reaches the wetland more often. Urbanization can also prevent infiltration through native soils into the shallow groundwater zone (Booth 1991, Azous and Horner 2001).
- The **timing** of water's presence and duration is changed by the use of engineered stormwater systems. Water is collected from impervious surfaces into stormwater ponds. Infiltration into shallow groundwater is prevented. The stormwater is discharged at given rates for longer durations into downstream receiving waters (Booth 1991, Azous and Horner 2001).
- The **frequency** of runoff volumes from storm events increases. The volume of runoff normally generated from small storm events is generated by smaller

volumes of precipitation (Booth 1991, Azous and Horner 2001, Reinelt and Taylor 2001, Thom et al. 2001).

- The **duration** of particular flows becomes extended as large volumes of stormwater are discharged at metered rates over longer periods of time (Booth 1991, Thom et al. 2001).
- The **rate of change** is increased through increasing the frequency and magnitude of water level fluctuations in urbanizing watersheds (Azous and Horner 2001, Reinelt and Taylor 2001, Thom et al. 2001).

3.4.4 Increased Input of Sediment Resulting from Urbanization

Researchers in the Puget Sound area have documented that urbanization increases erosion and this, in turn, increases the movement and deposition of sediment in wetlands (Booth 1991, Azous and Horner 2001, Reinelt and Taylor 2001, Thom et al. 2001).

Studies at the national level undertaken by the U.S. Environmental Protection Agency confirm that sediment in urban runoff is a problem nationwide (Tasker and Driver 1988). Sediments and solids constitute the largest volume of pollutant loads to receiving waters in urban areas (U.S. Environmental Protection Agency 2003).

A major source of sediment in urban areas comes from construction when the surface of the soils is disturbed and exposed to erosive forces. Runoff from construction sites is by far the largest source of sediment in urban areas under development (U.S. Environmental Protection Agency 1993).

3.4.5 Increased Input of Nutrients Resulting from Urbanization

Research on the impacts of urbanization in the Puget Sound area (Booth 1991, Azous and Horner 2001, Reinelt and Taylor 2001, Thom et al. 2001) has also documented that urbanization increases the amount of nitrogen entering aquatic systems including wetlands. Studies at the national level undertaken by the U.S. Environmental Protection Agency confirm that nitrogen in urban runoff is also a problem nationwide (Tasker and Driver 1988).

Nutrients are introduced into runoff from a number of different sources that include nutrients bound to sediment from construction sites, fertilizers applied to lawns, and decomposing grass clippings and leaves left on impervious surfaces (Johnson and Juengst 1997).

In addition to the application of fertilizers in residential areas, nitrogen is introduced into aquatic systems and wetlands from the release of nitrogen compounds in car and truck engines and through the burning of wood and coal (Paerl and Whitall 1999). The amount

of nitrogen coming from the deposition of these air pollutants in the United States is about 20% of the total excess nitrogen derived from human activities (Prospero et al. 1966). In heavily urbanized areas such as the Eastern Seaboard, the total amount of nitrogen coming from combustion can be as high as 40% or more of the total input by all human activities (Valigura et al. 1966).

3.4.6 Increased Input of Toxic Contaminants Resulting from Urbanization

In addition to sediment and nutrients, urban land uses generate a wide range of pollutants that include the following (U.S. Environmental Protection Agency 1993):

- Heavy metals (copper, lead, zinc)
- Hydrocarbons
- Organic matter that reduces oxygen
- Pesticides

Schueller and Holland (2000) cite a number of studies indicating that urban pollutant loads are directly related to the amount of impervious surface in the watershed. Impervious surfaces such as roads, parking lots, and storage yards are places where toxics from numerous sources collect. Precipitation falling on the impervious surfaces washes the collected chemicals and particles into the storm drain system (Schueller and Holland 2000).

The runoff from many different types of land use in urban areas can be toxic to aquatic life. Pitt et al. (1995) studied the relative toxicity of the runoff from different types of land uses in urban and suburban areas. Parking areas, storage areas, and landscaped areas (lawns, gardens) had the highest toxicity with approximately 20% of the samples being highly toxic. Over half of the samples of runoff from these urban land uses were moderately toxic.

Sriyaraj and Shutes (2001), working in London, documented that hard rains after extended dry periods result in the greatest concentrations of pollutants. This is also known to occur in Washington, where the greatest concentration of pollutants in surface runoff is typically observed in the fall with the first rains following summer drought (Booth 1991).

3.4.6.1 Heavy Metals and Hydrocarbons

Most heavy metals in urban runoff are adsorbed to sediment particles, although copper and zinc can occur in dissolved forms (Canning referenced in Newton 1989). The sources of heavy metals are various including motor vehicle brake linings, tire particles on roadways, emissions from vehicles, and industrial sources.

Sriyaraj and Shutes (2001) found that sediment from road runoff had high to moderate levels of heavy metals associated with it, and the metals were deposited within the sediments of the receiving wetland. Heavy metals, such as lead, zinc, copper, and cadmium, are some of the pollutants that accumulate on roads during dry summers. These pollutants are particularly concentrated when they are washed off during intense storms following long dry periods (Sriyaraj and Shutes 2001). Thurston (1999) found that lead and petroleum hydrocarbons were the most common pollutants attached to particles in an urban wetland receiving direct runoff from a municipal garage parking lot.

Most of the adsorbed metals are buried in sediment deposits within wetland substrates, thereby becoming substantially "locked up" from further biological activity (Canning in Newton 1989).

3.4.6.2 Organic Matter and Pesticides

Another contaminant present in runoff from urban areas is organic matter. As this organic matter decomposes in the water, it uses up oxygen that is dissolved in the water (called dissolved oxygen or DO). DO plays the same role as atmospheric oxygen in that it is critical for biological activity in aquatic communities. Oxygen is used by aquatic organisms. It is also used by bacteria for the decay of organic matter. This is called the biological oxygen demand (BOD) of the system. In natural systems, BOD fluctuates as oxygen use and organic inputs vary both daily and seasonally. The natural BOD of a system is thrown out of balance when there is excessive organic matter in the system. An increased BOD results in a decreased availability of dissolved oxygen.

Contaminants in urban runoff that cause increases in BOD include:

- Septic system effluent
- Oil and grease
- Pesticides or organic matter such as dog and cat feces
- Incidental sources from atmospheric fallout

Direct urban runoff can create a demand for oxygen that is equal to or greater than that from sewage effluent. BOD from urban runoff can have substantial cumulative effects (Canning, as referenced in Newton 1989).

3.4.7 Reduced Connectivity (Fragmentation of Habitat) Resulting from Urbanization

Urbanization causes fragmentation of habitat as new developments divide undisturbed areas (COST-Transport 2003). Conversion of the land for urbanization has turned large, continuous patches of habitat into numerous small patches, which are isolated from each other and surrounded by land uses that are not hospitable to native wildlife species (Aurambout 2003). The fragmentation of habitat continues to increase as the human

population grows (Dale et al. 2000). Developed lands in the U.S. increased by 18% between 1990 and 2000 to total 4.4% of the area of the country (Dale et al. 2000).

Wetlands, as part of an undisturbed landscape, are also subject to the fragmentation that results from urbanization. No specific information was found on the extent of fragmentation of wetlands resulting from urbanization.

3.4.7.1 Other Disturbances Resulting from Urbanization

Several other types of disturbances have been attributed to human activities in urbanizing areas:

- Alteration of soils
- Construction of roads
- Noise
- Recreational access
- Invasion by exotic plant and animal species

These disturbances are not discussed in detail in this chapter because little information was found describing how urbanization creates these disturbances. The impacts of these disturbances on wetlands have been documented and are summarized in Chapter 4.

3.4.8 Summary of Key Points

- Increases in urban population are generally accompanied by increased development density and sprawl. Wetlands in these areas may be converted to urban land uses or are degraded through a variety of causes.
- Urbanization results in clearing of vegetation, compaction of soil, modifications to water movement, alterations to riparian corridors, human intrusions, introduction of chemical contaminants, and increased areas of impervious surface. These changes profoundly affect environmental processes in contributing basins and, therefore, the downgradient drainage systems.
- Urbanization alters the movement of water into aquatic systems. Consequences of increased amounts of water include an increased frequency of erosive flows, greater volume of runoff, and longer duration of flows.
- The percent of impervious surface in the watershed and the size of the wetland are important in determining the amount of water level fluctuation that occurs as a result of urbanization.
- Urbanization may reduce shallow subsurface flow and base flow, but definitive studies on this topic are not available.

- With urbanization comes increased transport of sediment, nutrients, metals, oil, pesticides, and other contaminants in surface runoff.
- Fragmentation results as connections between wetlands and upland habitats are eliminated.

3.5 Disturbances Caused by Logging

In general, logging causes several types of disturbance that can impact the factors that control wetland functions and therefore affect the performance of those functions. These disturbances include (as reviewed in Cooke in press):

- Increased peak flows
- Increased water level fluctuations
- Increased nutrients
- Increased sedimentation
- Changes in soils
- Invasion by exotic species

The effects of logging in watersheds have recently received much attention. As a result, the scientific literature is being reviewed and synthesized by the Washington State Department of Natural Resources and is now in a draft form (Cooke in press). Therefore, this review of the literature does not cover the disturbances that result from logging and their impact on wetland functions.

3.6 Disturbances Caused by Mining

Surface mining generates large quantities of unusable rock that is often left on the surface after it is extracted. This exposes the rock (called spoils) to an oxidizing environment, resulting in a complex series of chemical reactions. The minerals contained in the spoils are not in equilibrium with the oxidizing environment and almost immediately begin weathering and mineral transformations.

The reactions are analogous to "geologic weathering" which takes place over extended periods of time (hundreds to thousands of years) but the rates of reaction are orders of magnitude greater than in "natural" weathering systems. The accelerated reaction rates can release damaging quantities of acidity, metals, and other soluble components into the environment. (U.S. Department of the Interior, Office of Surface Mining 2003.)

Thus, the two major disturbances created by mining are (Adamus et al. 2001):

- An increase in the levels of heavy metals that are toxic to many organisms
- An increase the acidity of surface waters

3.7 Chapter Summary and Conclusions

The focus of Chapter 3 has been to describe how different land uses may change the environmental factors that control wetland functions. A general conclusion that can be made from the scientific literature is that disturbances of environmental factors can occur at several geographic scales. Much of the early research focused on disturbances that occur at a single site or wetland. More recent research has documented the significance of disturbances that occur at the much larger scale of a watershed (called the landscape scale). The disturbances created by different land uses are summarized in Table 3-3 (at the end of this section) by the type of land use, the severity of the disturbance, and the scale at which the disturbance occurs. This table represents a synthesis of the severity of impacts by the authors of this document based on the information in the literature.

The effects of different land uses on the flow and fluctuations of water are well documented. Changes in land uses and vegetation communities on the land, whether for agriculture or as a result of urbanization, alter the patterns of surface and shallow groundwater movement across a landscape. Flows of water can be reduced or increased by different land uses as can the frequency and amplitude of water levels.

Removal of vegetation and/or compaction of native soils through agricultural practices, creation of lawns or grazed pastures, or creation of impervious surfaces all have the same relative consequence: increased volumes of water and rates of flows after a given storm event. The threshold of roughly 10% imperviousness within a basin appears to be the point above which significant impacts begin to occur to aquatic ecosystems based on research in the Puget Sound Basin.

While the effects of urbanization on water movement have been extensively studied, agriculture can also influence the water regime of wetlands, leading to loss of wetlands in some areas and creation or maintenance of wetlands in other areas where wetlands did not originally exist, such as areas influenced by irrigation.

Human activities also increase sediment and other pollutants in runoff. In agricultural areas, pesticides and fertilizers can contribute to contamination of surface waters. In urban areas, stormwater runoff frequently contains sediment, organic matter, phosphorus, metals, and other pollutants. Pollutants often adhere to sediment particles that enter wetlands. Mining increases the acidity of surface waters as well as adding toxic heavy metals. Logging increases sediments in a watershed and can also change the amount of water and its fluctuations.

Fragmentation of habitats is of increasing concern. As connections between wetlands and other habitats are broken and more wetlands across the landscape are converted to other uses, the remaining habitat becomes more isolated.

A key finding of this chapter is that different land uses may cause the same change in the controls of wetland functions. For example, urban land uses, agricultural practices, and logging have all been shown to increase sediments in a watershed. From the wetland's "point of view," the source of the sediment is irrelevant—the impact of excess sediments on wetland functions is similar, regardless of the source of sediments.

Chapter 4 shifts from a focus on the disturbances caused by human land uses (agriculture, urbanization, logging, mining) to describe how these disturbances impact wetlands and their functions.

Table 3-3. Disturbances resulting from different land use practices that can change the factors that control wetland functions.

Key to symbols used in table:

(xx) land use creates a significant disturbance of environmental factors

(x) land use creates a disturbance

(nm) studies on impacts of this land use do not mention this disturbance

(h) literature is lacking but disturbances can be hypothesized based on authors' experience

(?) information lacking

Disturbance	Scale of Disturbance	Agriculture	Urbanization	Mining
Changing the physical structure within wetlands (filling, vegetation removal, tilling of soils, compaction of soils)	Site scale	xx	xx	h
Changing the amounts of water	Landscape scale	xx	xx	?
	Site scale	xx	xx	h
Changing fluctuations of water levels (frequency, amplitude, direction of flows)	Landscape scale	xx	xx	?
	Site scale	xx	xx	h
Changing the amounts of sediment	Landscape scale	xx	xx	h
	Site scale	xx	xx	h
Increasing the amount of nutrients	Landscape scale	xx	xx	nm
	Site scale	xx	xx	nm
Increasing the amount of toxic contaminants	Landscape scale	xx	xx	x
	Site scale	xx	xx	xx
Changing the acidity	Landscape scale	nm	nm	x
	Site scale	nm	nm	xx
Increasing the concentrations of salt	Landscape scale	x	nm	nm
	Site scale	x	nm0	nm

Decreasing the connection between habitats	Landscape scale	xx	xx	h
Other disturbances	Site scale	xx	++	h

Chapter 4
Impacts of Human Disturbances on the Functions of Wetlands

4.1 Reader's Guide to This Chapter

Chapter 4 integrates the concepts discussed in Chapter 2 and Chapter 3. Chapter 2 described the functions performed by wetlands and the environmental factors that control these functions. Chapter 3 discussed the major disturbances caused by different human activities and uses of the land. This chapter continues by summarizing how each of the disturbances ultimately leads to impacts on wetland functions.

4.1.1 Chapter Contents

Major sections of this chapter and the topics they cover include:

Section 4.2, Introduction and Background on the Scale of Impacts to Wetland Functions describes how disturbances that impact functions in wetlands can occur either within the wetland itself or in the surrounding landscape. While the literature generally does not distinguish the scale of the disturbance when assessing impacts on wetland functions, there are some disturbances at the site scale that can remove all or most functions of the wetland (such as changing the physical structure of the wetland through filling).

Following this introduction, the chapter continues by describing how the major types of disturbances resulting from human activities affect wetland functions. As discussed in Chapter 3, different land uses may create the same type of disturbance (for example, both agriculture and urbanization may cause sedimentation). Therefore, each of the remaining sections of this chapter focuses on the different types of disturbances, without division by land use type, and their impact on each wetland function, as follows:

Section 4.3, Impacts of Changing the Physical Structure within a Wetland

Section 4.4, Impacts of Changing the Amount of Water in Wetlands

Section 4.5, Impacts of Changing the Fluctuation of Water Levels within a Wetland

Section 4.6, Impacts of Changing the Amounts of Sediment

Section 4.7, Impacts of Increasing the Amount of Nutrients

Section 4.8, Impacts of Increasing the Amount of Toxic Contaminants

Section 4.9, Impacts of Changing Acidity

Section 4.10, Impacts of Increasing the Concentrations of Salt

Section 4.11, Impacts of Decreasing the Connection between Habitats

Section 4.12, Impacts of Other Human Disturbances

Within each section, the impact of each disturbance is summarized in terms of the following wetland functions:

- Impacts on hydrologic functions
- Impacts on functions that improve water quality
- Impacts on plants
- Impacts on habitat for invertebrates
- Impacts on habitat for amphibians and reptiles
- Impacts on habitat for fish
- Impacts on habitat for birds (wetland-dependent species and wetland users)
- Impacts on habitat for mammals (wetland-dependent species and wetland users)

Section 4.13, Chapter Summary and Conclusions ties together the major concepts presented in the chapter.

4.1.2 Where to Find Summary Information and Conclusions

Each major section of this chapter concludes with a brief summary of the key points resulting from the literature on that topic in a bullet list format. The reader is encouraged to remember that a review of the entire section preceding the summary is necessary for an in-depth understanding of the topic.

For summaries of the information presented in this chapter, see the following sections:

- Section 4.3.9
- Section 4.4.9
- Section 4.5.9
- Section 4.6.9
- Section 4.7.9
- Section 4.8.9

- Section 4.9.9
- Section 4.10.9
- Section 4.11.9
- Section 4.12.6

In addition, Section 4.13 provides a summary and conclusions about the overarching themes gleaned from the literature and presented in this chapter.

4.1.3 Data Sources and Data Gaps

Data on some of the subjects related to the impacts of human disturbances on wetland functions are abundant for select areas in the state. For example, the Puget Sound Wetlands and Stormwater Management Research Program (Azous and Horner 2001) has provided numerous studies on how changes in land uses in a watershed affect the physical, chemical, and biological processes in wetlands of the Puget Sound lowlands.

Similarly, studies on the effects of changes in wildlife habitat resulting from physical changes within wetlands and reduced connection between habitats have been performed in Washington for some species and some types of habitat changes. The impacts to other species are less well studied or have only been examined in other states or other countries. Literature from other locales is included for these topics when relevant.

This chapter contains text that was adapted from a review of current scientific literature on the impacts of human activities on wetlands and their functions undertaken by the U.S. Environmental Protection Agency (Adamus et al. 2001). This review represents a very detailed summary of the literature published between 1990 and 2000 regarding wetlands across the United States. Portions of the review that were considered relevant to wetlands in Washington State were adapted for inclusion in this chapter, with permission from Dr. Adamus. The sections of this chapter that incorporate text adapted from the Adamus et al. (2001) review include:

- Section 4.3, Impacts of Changing the Physical Structure within a Wetland: Habitat for plants, invertebrates, reptiles and amphibians, fish, and mammals
- Section 4.4, Impacts of Changing the Amount of Water in Wetlands: Habitat for plants, invertebrates, reptiles and amphibians, fish, and birds
- Section 4.5, Impacts of Changing the Fluctuations of Water Levels within a Wetland: Habitat for invertebrates
- Section 4.6, Impacts of Changing the Amounts of Sediment: Habitat for plants and invertebrates
- Section 4.7, Impacts of Increasing the Amount of Nutrients: Habitat for plants, invertebrates, amphibians and reptiles, and birds

- Section 4.8, Impacts of Increasing the Amount of Toxic Contaminants: Habitat for plants, invertebrates, reptiles and amphibians, fish, and birds
- Section 4.9, Impacts of Changing the Acidity: Habitat for plants, invertebrates, reptiles and amphibians, and birds
- Section 4.10, Impacts of Increasing the Concentration of Salts: Habitat for invertebrates and birds
- Section 4.12, Impacts of Other Human Disturbances: Impacts to plant communities from altering soils, impacts of exotic invertebrates

The literature sources cited in the portions of the text that were adapted from the Adamus et al. (2001) report are included in the list of references at the end of Volume 1.

4.2 Introduction and Background on the Scale of Impacts to Wetland Functions

The disturbances that impact functions in wetlands can occur either within the wetland itself or in the surrounding landscape. Chapter 2 introduced the idea that the controls of wetland functions occur at both the "site scale" and the broader "landscape scale." As with the controls of wetland functions, disturbances caused by human activities can also occur at the same two scales (site and landscape).

For example, increased nutrients can flow into a wetland directly from an adjacent lawn or from animals grazing within the wetland (disturbance at the site scale). The nutrients could also originate from development or fertilizing fields somewhere higher in the contributing basin (disturbance at the landscape scale). As another example, the water levels in a wetland can be increased through the direct discharge of stormwater into the wetland (the site scale) or by adding impervious surface higher in the contributing basin (the landscape scale).

Much of the discussion in this chapter does not differentiate the scales at which the disturbance occurs. For example, the impacts on wetland functions resulting from excess nutrients or higher water levels can be expected to be the same whether they are delivered directly to the wetland or come from a distant source in the contributing basin. The literature does not usually differentiate between scales when discussing the impacts on wetland functions.

However, an alteration to the physical structure of the wetland itself is a type of disturbance that occurs only at the site scale. Filling, removing vegetation, tilling, or grazing within a wetland has a direct impact on the functions at that site. The most extreme impact to a wetland is the complete removal of all the factors that contribute to the existence of the wetland. Thus, filling a wetland or draining all the water eliminates all of the wetland functions because the wetland no longer exists.

4.3 Impacts of Changing the Physical Structure within a Wetland

Disturbances that directly change the structure of wetlands can be so severe that the wetland is destroyed. Filling or draining a wetland can so alter the water regime that the land can no longer support the wetland vegetation and maintain hydric soils. If a wetland is lost, most if not all of its wetland functions are also lost. Dahl (1990) estimated that 31% of the wetlands in Washington State had been lost prior to the 1980s as a result of filling or draining to the extent there is no longer enough water to maintain areas as wetland.

There are, however, some human alterations of the structure in wetlands that do not result in the complete loss of functions, including:

- Human removal of vegetation (for example, logging, mowing, or application of herbicides)
- Animal grazing
- Alteration of the soil through tilling or compaction
- Partial draining

This section describes what the literature says about how these alterations impact wetland functions. The impacts of grazing and removal of vegetation are better understood than those of alterations to the soils. Information was not available on how some of these alterations affect the wetland functions described in the following sections, and some impacts are hypothesized based on synthesizing other information.

4.3.1 Impacts of Changing the Physical Structure on Hydrologic Functions

No information was found on how changing the physical structure of wetlands impacts their hydrologic functions (reducing peak flows, reducing erosion, and recharging groundwater). One could hypothesize that removing erect and persistent vegetation (emergent, shrub, or forest species) may impact the reductions in water velocity that occur in wetlands. The density of vegetation is an important factor in reducing flooding or storm flows. If this vegetation is removed, the wetland will probably not be as effective at slowing these flows (in other words, there will be a change in how this wetland function is performed). As a result, downstream erosion and flooding may increase.

4.3.2 Impacts of Changing the Physical Structure on Functions that Improve Water Quality

No information was found on how changing the physical structure of wetlands affects how well wetlands remove pollutants. Removal of vegetation has impacts on both bacteria and plants, and this may affect the uptake and transformation of nutrients and toxic compounds in a wetland. The same can be hypothesized for direct alteration of soils, which may affect the chemical properties in a wetland. It is not possible, however, to predict or hypothesize how such changes might alter the wetland functions (that is, whether functions to improve water quality will increase or decrease).

4.3.3 Impacts of Changing the Physical Structure within Wetlands on Plants

By definition, removal of any vegetation causes at least a short-term change in plant biomass and possibly the composition of plant species. Vegetation can be removed by fire, tilling, mowing, or consumption of plants by animals including grazers (Newman 1991, Naiman and Rodgers 1997). Mortality from contaminants such as herbicides, logging or beaver activity, dredging or construction activities, or damage from wind, ice, or flooding can also cause loss of plants (Adamus et al. 2001).

The process by which vegetation is removed appears to influence the type, duration, and magnitude of the impact on plants. Impacts depend partly on the process through which the plants re-establish. When all or nearly all of the plants are removed through methods lethal to vegetation (such as with herbicides), recovery occurs mainly via recruitment of seeds. When removal is by non-lethal methods (such as grazing), recovery often is by vegetative regrowth. Vegetation patterns in some wetlands result in part from the differing causes of plant removal and whether those causes are lethal or not (Baldwin and Mendelssohn 1998).

The effects of grazing on wetland plants depend partly on the density of grazers, how long they are present in the grazed area, the availability of food and water in nearby alternative habitats, and the season (Clary 1995, Fitch and Adams 1998).

A study of riparian vegetation in eastern Oregon used different simulated grazing treatments to determine the effects of light and heavy grazing (Clary et al. 1996). While not clearly identified, it is evident that some of the plots were in riparian wetlands and others in non-wetland riparian habitats. The authors observed that herbaceous plants increased in growth and vigor for the ungrazed and moderately grazed plots, particularly if the grazing occurred only in the spring. Heavier, season-long grazing had detrimental effects on the vegetation.

In another study of riparian meadows in Oregon, Clary (1995) found that the biomass of the grass redtop (*Agrostis* sp.) remained stable or increased at a low-elevation site the year following simulated grazing treatments. At higher elevations, sedge species (almost all of which are found mostly in wetlands) either maintained or declined in biomass

production the following year. The author concludes that grazing only annually (for several months once a year as opposed to year-round) would significantly reduce sedge production, while not decreasing redtop production.

4.3.4 Impacts of Changing the Physical Structure on Habitat for Invertebrates

The presence of invertebrate species in a wetland is influenced by the type of plants that grow there. For example, in a Washington pond, some leeches (*Helobdella*), aquatic sowbugs (*Asellus*), mayflies, and some dragonflies (especially the large-bodied *Anax*) were more commonly associated with emergent vegetation than with submerged vegetation or open water areas. Midges, freshwater shrimp (*Hyalella azteca*), and mollusks (especially *Lymnaea* sp., *Gyraulus* sp., and *Anodonta* sp.) were more common on the submerged plants (Parsons and Matthews 1995).

The removal of vegetation either mechanically or through grazing, therefore, has a significant impact on the presence and abundance of invertebrate species in a wetland. Wetland managers often manipulate the structure of vegetation by mowing, burning, plowing, or planting to encourage or discourage populations of desirable or undesirable invertebrates (Batzer and Resh 1992, Kirkman and Sharitz 1994, de Szalay et al. 1996, de Szalay and Resh 1997).

Adamus et al. (2001) conclude from their literature review that the removal of vegetation:

- Removes substrates that would otherwise provide additional vertical space in the water column for invertebrates to colonize
- Removes shade, thus increasing water temperature and causing stress for invertebrates
- Increases the circulation and perhaps the velocity of water, with accompanying increases in dissolved oxygen and possible resuspension of sediments; this may result in changes to the habitats that favor different species of invertebrates
- Reduces inputs of leaf litter that provide food to some invertebrate taxa
- Reduces structures that otherwise shelter invertebrates from predators (Jordan et al. 1994)

Complete removal of vegetation generally reduces the richness of the wetland invertebrate community, but patchy removal or moderate grazing sometimes increases richness (McLaughlin and Harris 1990, Gray et al. 1999).

4.3.5 Impacts of Changing the Physical Structure on Habitat for Amphibians and Reptiles

The information on the impacts of direct disturbances to the physical structure of a wetland on amphibians is ambiguous for Washington. In the Puget Sound Basin of Washington, surveys of 19 wetlands found no correlations that were statistically significant between amphibian richness and vegetation form (Richter and Azous 1995). Plant stem diameter is apparently more important than plant species (Richter 1997). For example, stems less than 0.1 inch (3 mm) diameter were preferred by the northwestern salamander (*Ambystoma gracile*) regardless of the plant species (Richter and Roughgarden 2002). Thus, impacts to amphibians from selective cutting or harvesting cannot be predicted.

The density of submerged plants is also important. A survey of 40 wetlands in the Puget Sound area found more native species of amphibians among wetlands containing dense emergent vegetation (Adams and Bury 1998). Dense vegetation may help protect the larvae of native aquatic amphibians from larger predators. It can be hypothesized, therefore, that removing dense emergent vegetation would probably impact the populations of amphibians.

Other studies have focused on the impacts of grazing. Based on personal observations, Maxell (2000) asserts that livestock grazing can impact amphibians through:

- Direct trampling of animals
- Trampling of vegetation that results in loss of habitat and reduces insect populations that are food sources for amphibians
- Contamination of water bodies through livestock waste
- Changes in substrate composition and bank structure
- Increased sedimentation

However, a contradictory study of the Columbia spotted frog in 127 ponds in northeastern Oregon found no significant differences between grazed and ungrazed ponds in terms of the numbers of frog egg masses and the abundance of recently metamorphosed frogs (Bull and Hayes 2000). Egg mass volume was larger at grazed sites, possibly due to a greater presence of adults or an older population (older, larger females lay bigger egg masses). Six of the eight most productive ponds (those with 20 or more egg masses) were grazed, indicating that grazing had no detrimental effect on this frog in grazed wetlands.

4.3.6 Impacts of Changing the Physical Structure on Habitat for Fish

Information in the literature did not differentiate between resident and anadromous fish. However, it does address fish in general. For example, the removal of vegetation can have a significant impact on the fish present in a wetland as a result of (Adamus et al. 2001):

- Increased water temperature that may go above the tolerance limits of certain species
- Decreased cover and thereby increased susceptibility to predation
- Changes in foods and their availability

Woody material is especially important as a source of cover for fish in off-channel wetlands such as oxbows and sloughs and in lakes (Leitman et al. 1991, Dewey and Jennings 1992, Fausch and Northcote 1992, McIntosh et al. 1994).

In lacustrine fringe wetlands, submerged plants are particularly important and their removal can change the habitat for fish. For example, declines in plants resulting from introductions of grass carp (Bain 1993) have been linked to an increase in the proportion of limnetic or open water fish species (Bettoli et al. 1991, Maceina et al. 1991, Martin et al. 1992). However, intentional thinning of plant beds can sometimes result in higher growth rates of some age classes of lake fish, presumably by giving them better access to invertebrates that are their food source (Olson et al. 1998).

4.3.7 Impacts of Changing the Physical Structure on Habitat for Birds

Many birds are sensitive to the presence and type of vegetation and its location in relationship to open water (Kauffman et al. 2001). The removal of vegetation is therefore expected to change the distribution and abundance of birds in wetlands.

Grazing has also been found to change the distribution of birds. In a study in southeastern Oregon on the effects of grazing on birds, researchers used exclosures to remove livestock from portions of riparian meadows (Dobkin et al. 1998). They found that the richness and abundance of bird species increased within the exclosures in comparison to the plots that remained available for livestock grazing. Moreover, the exclosures were dominated by wetland and riparian birds while the open plots were dominated by upland bird species.

A study of riparian habitats in Montane areas of Nevada suggests that grazing reduces the amount of suitable habitats for nesting riparian bird species (Ammon and Stacey 1999). These authors concluded that grazing reduces streamside vegetation and the diversity of vertical structure, thus making suitable nesting substrates less available. By placing

artificial nests in both areas, they found that grazing facilitated nest predation, possibly because nests were more easily detected by predators in the grazed area, or because there were different predator species on each site. This study relied on artificial nests for much of the data presented on nesting success, which the authors note may be problematic. There also is a lack of sample replication in that only one area was studied for each of the two treatments.

The changes in the structure of vegetation that result from the conversion of forested wetlands to emergent and open water wetlands can alter species composition and richness of breeding birds. For example, 53% of the bird species that formerly used forested wetlands no longer occur regularly where such forests have been logged and converted to emergent wetlands (Doherty 2000 as reported in Adamus et al. 2001).

4.3.8 Impacts of Changing the Physical Structure on Habitat for Mammals

Many mammals are sensitive to the presence and type of vegetation and its location in relationship to open water. The removal of vegetation is therefore expected to change the distribution and abundance of mammals in wetlands (Adamus and Brandt 1990).

Adamus and Brandt (1990) created a synthesis of the literature on mammal habitat which serves as the basis for the following discussion.

The species richness of small mammals in wetlands has been correlated with the complexity of vegetation structure (Arner et al. 1976, Landin 1985, Nordquist and Birney 1980, Stockwell 1985, Searls 1974, Simons 1985). Removal of vegetation and associated long-term destruction of den sites in both wooded and emergent wetlands have caused changes in furbearer populations and small-mammal communities (Krapu et al. 1970, Malecki and Sullivan 1987). In contrast, restoration of riparian vegetation has led to increases in use by mink (Burgess and Bider 1980).

Grazing at levels recommended by the Natural Resources Conservation Service had no significant effect on the abundance or distribution patterns of small mammals in a cottonwood floodplain in Colorado (Samson et al. 1988).

4.3.9 Summary of Key Points

- Filling or draining a wetland can so alter the water regime that the land can no longer support wetland vegetation and maintain hydric soils. If a wetland is lost, most if not all of its functions are also lost.
- Some direct disturbances of wetlands, such as removal of vegetation, grazing, and alteration of the soil, change the wetland functions but do not result in the complete loss of functions.

- Impacts of removing vegetation on the habitat functions in wetlands have been documented for invertebrates, fish, birds, and mammals. Impacts on amphibians, however, are ambiguous. Impacts to the hydrologic and water quality functions resulting from vegetation removal can only be hypothesized since no information was found in the literature.
- Impacts of grazing on habitat functions have been documented for invertebrates and birds and are somewhat conflicting for amphibians. Impacts to fish habitat have not been studied. The one study of mammals suggests that low levels of grazing in a floodplain may have minimal impacts on the habitat of this group. No information was found on impacts of grazing on the hydrologic and water quality functions.
- No information was found on the impacts of soil alterations (through tilling and compaction) on any of the functions performed by wetlands.

4.4 Impacts of Changing the Amount of Water in Wetlands

4.4.1 Impacts of Changing Amounts of Water on Hydrologic Functions

Specific documentation was lacking on how increasing or decreasing amounts of water may affect wetland functions in reducing flooding or erosion or recharging groundwater. It can be hypothesized, however, that the storage capacity of a wetland in a depression during floods will be reduced if water levels increase. The volume that would have been available to store floodwaters is used instead to store the increased volumes coming into the wetland. This suggests that the functions related to reducing flooding would also decline because storage is a large component of flood reduction. On the other hand, wetlands in which water is deeper or covers more of the wetland may provide better recharge of groundwater because infiltration depends on the depth of water in the wetland (hydraulic head) and the area that is submerged (Hruby et al. 1999).

The converse can be hypothesized if water levels in wetlands decrease. The potential amount of water that can be stored in a wetland will increase as it becomes drier, thereby increasing the "flood reduction" functions. The function of recharging groundwater would decrease because less water would be present and it would be shallower.

4.4.2 Impacts of Changing Amounts of Water on Functions that Improve Water Quality

Some information suggests that flooding of wetlands (increased amounts of water) stimulates microbial activity, and this in turn may change how a wetland removes pollutants.

The activity of microbes potentially increases conversion of inorganic mercury to the much more toxic methyl mercury form (Kelly et al. 1997). In this case flooding would reduce the effectiveness of a wetland at improving water quality because the wetland may become a source of this more toxic compound.

Increased amounts of water may also have an impact on denitrification in wetlands. Adamus et al. (2001) reviews several studies in which the water content of soil was found to be the dominant factor controlling denitrification. In Washington, the area that is seasonally inundated was judged to be a critical factor in determining denitrification (Hruby et al. 1999). If the increase in water levels expands the area that is seasonally flooded, denitrification rates will probably increase. If, however, increases in the amount of water in a wetland expand the amount of permanent water at the expense of the areas that were seasonally flooded, the rates of denitrification can be hypothesized to decrease. Thus, wetlands in which the water regime has been changed will probably have a different rate of denitrification than they had previously. The data are insufficient, however, to predict whether denitrification rates will be higher or lower, and the change in functions depends on how the water regime is altered.

4.4.3 Impacts of Changing Amounts of Water on Plants

Much of the literature on how changing amounts of water affect plant populations in wetlands of the Pacific Northwest is in terms of changes in the dynamics of water movement (hydroperiod). This concept combines both changes in water levels and changes in how water levels fluctuate (the latter is addressed as a separate disturbance in Section 4.5.)

The composition and richness of the plant community are influenced by the saturation in the root zones of wetland plants. This is influenced by:

- The **duration** of saturation (Dicke and Toliver 1990, Merendino and Smith 1991, David 1996, Vivian-Smith 1997, Silverton et al. 1999)
- The **timing** of saturation (Merendino et al. 1990, Squires and van der Valk 1992, Scott et al. 1996, 1997, Gladwin and Roelle 1998)
- The **frequency** of saturation (van der Valk 1994, Pezeshki et al. 1996, 1998, Smith 1996, Pollock et al. 1998)

Disturbances to the dynamics of water movement and volume in a wetland can cause major changes in the distribution and richness of plant species. The response of an individual wetland to such changes, however, is difficult to predict. The existing information indicates that each plant species responds in a different way to changes in water levels. This means that overall the response of the plant community in a wetland will depend on the sum of the responses of the individual species. The following discussion summarizes some of the studies documenting how plant communities change with changes in water levels. It is beyond the scope of this document to provide detailed information on the response of individual plant species.

Responses of hundreds of plant species to specific hydrologic variables that have been studied are presented in a database at EPA's web site (Adamus and Gonyaw 2000). The database is available at http://www.epa.gov/owow/wetlands/bawwg/publicat.html

The changes in plant communities are linked to differences among plant species in their ability to resist drought and flooding. The life history and physical characteristics of plants play a role (Earnst 1990, Koncalova 1990, Voesenek et al. 1993, Kirkman and Sharitz 1993, Teutsch and Sulc 1997). The characteristics of seed dispersal and germination of plants relative to water dynamics may have the greatest effect on the relative abundance of species, according to a simulation conducted by Ellison and Bedford (1995) using six years of data from a southern Wisconsin sedge meadow. Some species, such as cattail (*Typha* spp.), are able to keep pace with rising water levels because their stem tissue elongates rapidly and to a greater degree than other species (Waters and Shay 1992, Galatowitsch et al. 1999) or they sprout adventitious roots (Voesenek et al. 1993).

Increases in inundation may change the exposure of plants to competitors and herbivores (Wilson and Keddy 1991) and cause a shift in the location of plant communities within a wetland (van der Valk et al. 1992). The opposite extreme—dehydration—kills plants partly by removing the pathway for taking up nutrients and maintaining tissues. Dehydration may also increase or decrease competition and plant exposure to herbivores (Adamus et al. 2001).

Woody plants are particularly sensitive to prolonged inundation, especially for longer than 80 days (Niswander and Mitsch 1995, Toner and Keddy 1997, Sharitz and Gresham 1997). Their seedlings consequently are most affected during years when flooding occurs at or shortly after the beginning of the growing season, or when flooding persists for more than 40% of the growing season (Toner and Keddy 1997). Annual (as opposed to perennial) species tend to increase proportionately in response to drought and some other severe disturbances (Poiani and Johnson 1989).

Species with small, light seeds seem particularly adept at colonizing mudflats exposed during drawdowns and after disturbances (Poiani and Johnson 1989, Ellison and Bedford 1995). These species tend to emerge early in the season and may be more successful by taking advantage of greater light availability (Toner and Keddy 1997).

Successive years of annual drawdowns can favor the spread of many non-native plant species within wetlands (van der Valk 1994). Dominance of a wetland by just a few species is sometimes a sign that the wetland has experienced prolonged drought or drawdown (Wilcox 1995).

Many species have only a narrow "window" in which they can germinate. For example, there may be only a few weeks when favorable water levels or a temporary lack of competitors must coincide with favorable temperatures and acceptable water quality (Rood et al. 1998).

4.4.4 Impacts of Changing Amounts of Water on Habitat for Invertebrates

Disturbances to the amount of water in a wetland can cause major changes in the distribution and richness of invertebrate species. Because each species responds in a different way to increases or decreases in water regime, the overall response of the invertebrate community in a wetland will depend on the sum of the responses of the individual species.

In general, the amounts of water in a wetland influence the distribution and richness of invertebrates by:

- Altering the amount and pattern of horizontal and vertical habitat space available for colonization (Adamus et al. 2001)
- Changing the types of algae and vascular plants that occur, the proportions of these two major food sources for invertebrates, and the seasons in which they occur (Murkin et al. 1991)
- Changing the extent of contact between plants and water, thus influencing attachment space, availability of detrital foods, shade, and shelter (Ross and Murkin 1993, De Szalay et al. 1996)
- Influencing the access of predators (Reice 1991, Martin et al. 1991, Mallory et al. 1994, Johnson et al. 1995, Wellborn et al. 1996)
- Affecting the intensity of competition (Wissinger et al. 1999)
- Causing mortality if complete desiccation or freezing occurs (Layzer et al. 1993)

4.4.4.1 Impacts of Reduced Amounts of Water on Habitat for Invertebrates

Some of the most dramatic changes to wetland invertebrate communities occur when wetlands that seldom or never dry out completely are subjected to drought or complete drawdown (Adamus et al. 2001). Less dramatic changes to invertebrate communities occur with slight alterations in the timing, duration, predictability, and depth of surface water (Eyre 1992, Giberson et al. 1992).

Drought and drawdown render the less mobile species of invertebrates more vulnerable to predation, as well as causing their direct loss due to desiccation and related factors (e.g., Stanley et al. 1994). Drought also seems to favor non-insect invertebrates, which can increase at the expense of the insect component of the invertebrate community (Hershey et al. 1999).

Coupled with the studies that show invertebrate richness increasing with longer periods of inundation, these observations indicate that removing water from a wetland may reduce the species richness of invertebrates.

4.4.4.2 Impacts of Increased Amounts of Water on Habitat for Invertebrates

An increase in the amount of water in a wetland seems to change the composition of the invertebrate community. Densities of swimming (nektonic) and bottom-dwelling (benthic) predatory invertebrates do not increase with flooding as much as the numbers of nektonic and benthic herbivores and detritivores. Predatory species can even decrease after flooding (Murkin et al. 1991), and they often increase as drought or drawdown progresses.

Although flooding generally increases the density and richness of invertebrates in wetlands, the increase may be short-lived. For example, flooding of Manitoba marshes (Murkin et al. 1991) to a level 3 feet (1 m) above normal caused a major increase in numbers of nektonic invertebrates in both vegetated and open water areas for only one year. Furthermore, densities of benthic invertebrates increased in flooded vegetation but not in open areas. The biomass of nektonic invertebrates increased only in the vegetated areas (Murkin et al. 1991).

Some researchers have observed that food webs become more complex and taxa numbers increase as wetlands become wetter, such as those that are ponded for longer periods. This has been observed in seasonal wetlands of eastern Washington (Lang 2000). Also, the use of emergence traps in 19 wetlands in King County yielded more taxa from permanently flooded than seasonally flooded wetlands (Ludwa and Richter 2000), suggesting that wetlands in which the water levels fluctuate more often will have fewer invertebrate species.

These results suggest that disturbances that cause water to remain longer in a wetland will probably increase species richness at first. The long-term effects of such increases, however, are not well understood.

4.4.5 Impacts of Changing Amounts of Water on Habitat for Amphibians and Reptiles

Most amphibians cannot tolerate prolonged dry periods. Drying of seasonal pools, especially when it occurs ahead of normal seasonal schedules, can greatly diminish the breeding success of amphibians (Rowe and Dunson 1993). This is partly because many amphibian species disperse only short distances (Berven and Grudzien 1990). Amphibian populations scattered across wetlands of varying depth and water permanence can enable species to survive long-term droughts or floods. The availability of numerous scattered wetlands can protect amphibians against effects of localized drought. Some frog and toad species living in relatively intact landscapes seem mostly unaffected, at the level of populations, by significant periods of drought (Dodd 1995).

However, amphibian populations recover slowly or not at all from droughts they might otherwise survive when habitat has become fragmented (see Section 4.11). Fragmentation results when wetlands are altered and the distance increases between remaining wetlands that are free of fish and most suitable for amphibians. Amphibian dispersal routes can be disrupted by construction of roads or other unsuitable habitats that displace terrestrial vegetation (Pounds and Crump 1994).

Both prolonged desiccation and extreme floods can increase opportunities for invasion of wetlands by exotic plant species. This can impact the suitability of a wetland as habitat for amphibians. Patterns of vegetation typically become more homogeneous, prey abundance may decline, and the habitat may become less suitable for amphibians (Ludwa 1994).

During a two-year drought in Washington, a local population of painted turtle (*Chrysemys picta belli*) suffered a 70% decline (Lindenman and Rabe 1990). This appeared to be due to both mortality and movement of turtles out of the wetland. Growth was suppressed but recovered as conditions improved. Drawing down the water level in the autumn to allow wetland management, flood control, or for other reasons can cause high mortality among juvenile overwintering turtles due to freezing if the drawdowns follow abnormally high water levels in late summer that attracted turtles (Galat et al. 1998).

These results indicate that changing the amounts of water in a wetland affects both amphibians and reptiles (specifically painted turtles). Impacts may occur from lowering the water levels (for example, through ditching, draining, or pumping) or raising the levels through increased flooding as a watershed is developed.

4.4.6 Impacts of Changing Amounts of Water on Habitat for Fish

Declines in the amounts of water alter the community structure of wetland fish. Fish experience a greater need to use overlapping resources and face an increased risk of predation when wetlands become drier (Adamus et al. 2001). Low water also increases the chance of fish freezing in winter or dying from thermal stress in summer (Adamus et al. 2001).

Sustained drawndowns can also reduce competition among fish that return to wetlands when water levels rise again by temporarily eliminating larval dragonflies and other large invertebrates that normally compete for food with the fish or prey on larval fish (Travnichek and Maceina 1994).

Impacts of increasing water levels on fish in wetlands were not documented in the literature.

4.4.7 Impacts of Changing Amounts of Water on Habitat for Birds

Disturbances to the amounts of water in a wetland can cause major changes in the distribution and species of birds. As with plants and invertebrates, the overall response of the bird community in a wetland will depend on the sum of the responses of the individual species.

4.4.7.1 Impacts of Reduced Amounts of Water on Bird Habitat

Drainage and some other disturbances in the amounts of water in wetlands have been well documented as contributing to the decline of many wetland bird species (David 1994, DeAngelis et al. 1997). In Manitoba, for example, wetland drainage has made breeding and brood-rearing areas for waterfowl less available (Rotella and Ratti 1992). As wetlands are drained or converted to other land cover types, local densities of wetlands decline and the average distances between individual wetlands increase.

Drought conditions also expose duck nests to greater predation. With drought, plants are less dense and vigorous, and islands that formerly were inaccessible gain new access points (Hallock and Hallock 1993, Jobin and Picman 1997).

Widespread drawdown of water tables reduces the number and perhaps the variety of wetlands and their vegetation communities. This in turn diminishes the richness, density, and breeding success of birds in many individual wetlands and wetland complexes (Higgins et al. 1992, Bethke and Nudds 1993, Bancroft et al. 1994, Greenwood et al. 1995, Dobkin et al. 1998).

4.4.7.2 Impacts of Increased Amounts of Water on Bird Habitat

Increasing the duration of saturation or inundation can change the use of wetlands by a variety of birds. This change can occur when shallow ephemeral ponds are dredged to make areas with longer periods of standing water (such as stock ponds). In the Columbia Basin, Creighton et al. (1997) found an increase in use by several species of diving and dabbling ducks, coots, and terns when shallow, densely emergent wetlands were dredged to create deeper pools of open water. They also documented an increase in the biomass of zooplankton, a food source for several guilds of wildlife. However, there was a decrease in use by sora (*Porzana carolina*) and Virginia rails (*Rallus limicola*) as well as red-winged blackbirds (*Agelaius phoeniceus*). The use of the excavated habitats by rails was expected to increase over time as emergent vegetation became reestablished in the excavated pools because rails prefer vegetation that is a mix of robust and thin-stemmed species. An increase in use by shorebirds was one short-term benefit. The shorebirds fed on the moist, fresh dredge spoils and exposed unvegetated soils of the newly excavated basins. Once the soils became vegetated, use by shorebirds declined.

On the other hand, while construction of reservoirs raises water levels, this affects birds by eliminating many wetlands through flooding and destabilizing water levels in the

remaining wetlands (Nilsson and Dynesius 1994). Associated changes in river morphology influence the species composition of wintering waterfowl (Johnson et al. 1996).

4.4.8 Impacts of Changing Amounts of Water on Habitat for Mammals

Documentation on how disturbances to the amount of water in a wetland may affect their ability to provide habitat for mammals was not found.

4.4.9 Summary of Key Points

- Impacts of reducing water levels on the habitat functions of wetlands have been documented for invertebrates, fish, birds, and amphibians. All these groups have reduced species richness and abundance when wetlands dry up.
- Impacts of increasing water levels in wetlands on its functions as habitat have been documented for invertebrates and birds. The species richness of invertebrates may increase for a short time if a wetland becomes wetter. The impacts on the populations of birds are mixed. In some cases the richness of birds increases and in some cases it decreases.
- Impacts to the suitability of wetlands as mammal habitat resulting from either increasing or reducing water levels have not been studied.
- Reducing the amount of water changes the distribution of plants in a wetland, but the studies did not address if species richness will increase or decrease. Data suggest that woody species will tend to be replaced by more grass-like species when water levels in a wetland increase.
- Impacts to the hydrologic and water quality functions from either increasing or reducing water levels can only be hypothesized since no information on these topics was found in the literature.

4.5 Impacts of Changing the Fluctuation of Water Levels within a Wetland

A major finding of the Puget Sound Wetlands and Stormwater Management Research Program was that fluctuations in water level are key in determining biological responses. There are different types of fluctuations in water levels in a wetland and these are described in the shaded box on the following page. The researchers found a decline in the biotic diversity of wetlands associated with an increase in water level fluctuations caused by expanding impervious area within the contributing basin (Reinelt et al. 1998, Azous and Horner 2001).

Prolonged inundation (that is, less frequent water level fluctuations) resulting in a lack of oxygen in the soils has been indicated as a factor in changing the biota of wetlands. Although many hydric soils may be anaerobic, changing the length of time the soils are inundated results in prolonged anaerobic conditions and chemical changes in the soils. These changes in soil chemistry influence the survival of vegetation and soil biota that were adapted to shorter periods of inundation (Thom et al. 2001). On the other hand, key habitat elements are eliminated and biotic diversity declines in wetlands with increased periods of summer drying (Azous et al. 2001).

Mechanisms for how water level fluctuations affect aquatic systems

Richter et al. (1996) developed a method to model "indicators of hydrologic alteration" based on assessing changes in 32 hydrologic parameters they identified as being relevant to the biotic integrity of aquatic ecosystems. They divided the parameters into the following five fundamental factors that characterized how fluctuations in water levels influence biotic communities in aquatic systems:

Magnitude is a measure of the availability or suitability of habitat. It defines such habitat attributes as wetted area or habitat volume, or the position of a water table relative to wetland or riparian rooting zones.

Timing is the timing of occurrence of a particular water condition. It can determine whether certain life-cycle requirements are met. It can also influence the degree of stress or mortality associated with extreme water conditions such as floods or droughts.

Frequency refers to the frequency of occurrence of specific hydrologic conditions, such as droughts or floods. It may be tied to reproduction or mortality events of various species, thereby influencing population dynamics.

Duration is the length of time over which a specific hydrologic condition exists. It may determine the success of a particular species' life cycle or the accumulation of stressful effects.

Rate of change in hydrologic conditions may be linked to stranding of individuals (in isolated pools or along a wetted edge). It may also be related to the ability of sensitive species to maintain root contact within the phreatic zone (the portion of the soil that is influenced by proximity to the groundwater table).

4.5.1 Impacts of Changing Fluctuations in Water Levels on Hydrologic Functions

The literature did not provide explicit information on possible impacts of changes in water level fluctuations on the hydrologic functions of wetlands. It is not possible at this stage to hypothesize either positive or negative impacts on hydrologic functions. The major questions that need to be addressed include:

- Will changes in the frequency or amplitude of water level fluctuations change the flood storage capacity of a wetland?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland reduces water velocity?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland recharges groundwater?

4.5.2 Impacts of Changing Fluctuations in Water Levels on Functions that Improve Water Quality

How changing fluctuations in water levels impact the ability of wetlands to improve water quality was not detailed in the literature. It is not possible to hypothesize either positive or negative impacts on water quality functions. The major questions that need to be addressed include:

- Will changes in the frequency or amplitude of water level fluctuations change how a wetland traps sediment?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland removes nitrogen?
- Will changes in the frequency or amplitude of water level fluctuations change the way in which a wetland captures or transforms toxic compounds?

4.5.3 Impacts of Changing Fluctuations in Water Levels on Plants

In general, the amplitude and rate of water level fluctuation have been found to influence the species composition, biomass, and germination of plants (Hudon 1997, Shay et al. 1999). Furthermore, the timing of inundation and duration throughout the seasons also influences plant species richness and survival (Ewing 1996, Reinelt et al. 1998, Owen 1999, Azous et al. 2001).

Researchers consistently found a decline in plant species richness in urbanized watersheds where water level fluctuations had increased (Azous and Cooke 2001).

Among 26 wetlands in the Seattle area, the degree of seasonal water level fluctuation was negatively associated with richness found in emergent and shrub wetlands, but it had no statistically significant effect on species richness in the forested wetlands (Cooke and Azous 2001). These authors found that fluctuation during the early spring seemed to have an especially detrimental effect on plant richness in the emergent and shrub wetlands.

Reinelt et al. (1998) found that the development of plant communities in lowland wetlands of Puget Sound was related to water level fluctuations and depth of inundation during the early growing season. They noted that shifts in the "hydrologic profile" of the wetland caused a subsequent shift in the species composition of the wetland's plants. The emergent and scrub-shrub communities of the wetland tended to have lower plant richness when average annual water level fluctuations increased to over 8 inches (20 cm).

Azous and Horner (2001) determined that the duration of flooding, as well as depth, also strongly influenced plant diversity. They noted greatest plant diversity when:

- Flooding events were less than 0.5 feet (0.2 m) above predevelopment levels
- Floods were limited to an annual average of three or fewer events per month
- The cumulative duration of flooding was less than six days per month above predevelopment averages

On the other hand, a lack of water level fluctuation can be just as damaging as excessive fluctuation to some wetland species (Rood and Mahoney 1990). This is because many species need a period of desiccation in order to germinate.

Evidence from some studies suggests that the relative tolerance to water level fluctuations is greatest among several non-native or invasive species (Figiel et al. 1991, Haworth-Brockman and Murkin 1993, King and Grace 2000). Increases in water level fluctuations and duration of inundation favor generalist plants (plants that are found under a wide range of environmental conditions) in the Pacific Northwest (Azous et al. 2001).

These results indicate that changes to water level fluctuations in wetlands will change the plant species present in the wetland. Furthermore, increases in water level fluctuations will probably facilitate the invasion of non-native or "aggressive" native species.

4.5.4 Impacts of Changing Fluctuations in Water Levels on Habitat for Invertebrates

In the Northwest, researchers have observed a decline in the diversity of invertebrates with an increase in impervious area in the basin, which to a large degree results in changes in the fluctuations of water levels (Ludwa 1994, Hicks 1996, Ludwa and Richter 2001a, Thom et al. 2001). Information from other parts of the United States seems to confirm this.

The densities of some invertebrate species can be decimated by rapid water level fluctuations, especially when the fluctuations are more frequent and severe than those historically encountered in the wetland. For example, Missouri floodplain pools that experience fluctuations in water level at extreme frequencies and amplitudes tend to have lower invertebrate density (Magee et al. 1993). Repeated exposure to desiccation in a short period of time can lead to a marked reduction in the density of invertebrates. In an Arizona stream that experienced 12 flash floods between August and December of a single year, densities of all invertebrates were reduced by 75 to 100% (Boulton et al. 1992). In particular, the numbers of water spiders, midges, and some caddisflies, mayflies, and snails declined.

In contrast, some groups of invertebrates appear quite resilient to periodic fluctuations. In the Arizona study referenced above (Boulton et al. 1992), the oligochaete (worm) populations appeared to be unaffected. In a British Columbia river, populations of the mayflies *Rhithrogena* and *Baetis*, as well as the caddisfly *Hydropsyche* (species found in wetlands as well) survived flows that increased rapidly during flooding from 500 to 6,500 cubic meters per second (Rempel et al. 1999). In an Oklahoma intermittent stream where spring and fall floods reduced the density of invertebrates by 90%, the mayflies *Caenis* sp., *Leptophlebia* sp., and *Baetis* sp. were especially resilient and midges were less so (Miller and Golladay 1996).

A number of studies have found that reducing fluctuations in streams by maintaining minimum water levels (such as in reservoirs) can increase invertebrate densities in the part of an adjacent wetland that is not permanently inundated (Weisberg et al. 1990, Troelstrup and Hergenrader 1990).

4.5.5 Impacts of Changing Fluctuations in Water Levels on Habitat for Amphibians and Reptiles

In Puget Sound wetlands, amphibian species richness was negatively correlated with the percent of impervious cover in a contributing basin. The primary cause is increased water level fluctuation (Richter and Azous 2001a). The richness of amphibians declined to less than three species when water level fluctuations increased to over 8 inches (20 cm) (Richter and Azous 2001a, Thom et al. 2001). Chin (1996) concluded that the reduced richness of amphibians was correlated with a reduction in the diversity of wetland plants that resulted from increases in water level fluctuations.

Increases in fluctuation of water levels also affect amphibians by (1) stranding egg masses when water levels drop, and (2) reducing the thin-stemmed emergent plant species on which amphibians lay their eggs. Previous unpublished work by Richter and Roughgarden (2002 in press) in western Washington found that amphibians preferred thin-stemmed vegetation on which to lay their egg masses. Greater water level fluctuation directly affects amphibian egg survival and causes changes in plant species, reducing the thin-stemmed emergent species used by amphibians for egg laying (Chin 1996, Richter and Roughgarden 2002 in press).

No correlations were found between the richness of amphibian species and a variety of other factors including wetland size, distance to breeding habitats, presence of predators, and number of vegetation classes (Richter and Azous 2001a). The most significant factor affecting species richness was mean water level fluctuation, with 8 inches (20 cm) mean annual fluctuation being a threshold for lentic breeding species (those that breed in stagnant or slow-moving waters such as ponds and wetlands). Lentic breeding amphibians appear to be affected by increases in the duration and frequency of flooding and increased discharge rates resulting from the greater frequency and magnitude of storm peaks in urban watersheds (Richter and Azous 2001a).

Amphibian populations in western Washington generally experience impacts in contributing basins with more than 10% impervious surface area (Booth and Reinelt 1993). A more recent study documented that watersheds with less than 15% total impervious area had three or more amphibian species, whereas most watersheds with more than 25% impervious area had less than three species (Chin 1996). Chin (1996) concludes that changes in water level fluctuations and maximum water levels during spring breeding and embryo development are the primary adverse effects of increased impervious surface.

4.5.6 Impacts of Changing Fluctuations in Water Levels on Habitat for Fish

Researchers compared the use of two watersheds in King County by two species of fish, coho salmon (*Oncorhynchus kisutch*) and cutthroat trout (*Salmo clarki*). They identified a "marked degradation" in relative fish use at 8 to 10% total impervious area within the watershed (Lucchetti and Fuerstenberg 1993 as cited in Booth and Reinelt 1993). It can be assumed that much of this impact is a result of changes in water level fluctuations because this is one of the major impacts of impervious surfaces in a contributing basin.

4.5.7 Impacts of Changing Fluctuations in Water Levels on Habitat for Birds

General observations have indicated a decline in bird richness for wetlands located in a contributing basin that is developed or developing. Richness was not reduced in contributing basins that remained rural or relatively undeveloped over the course of the Puget Sound Wetlands and Stormwater Management Research Program (Richter and Azous 2001b, Thom et al. 2001). However, these observations have not specifically been correlated with changes in the fluctuation of water levels.

4.5.8 Impacts of Changing Fluctuations in Water Levels on Habitat for Mammals

No explicit information on how changing fluctuations in water levels will impact mammal populations in wetlands was presented in the literature. It is not possible to hypothesize either positive or negative impacts on mammal populations.

4.5.9 Summary of Key Points

- No information was found on the impacts to the hydrologic and water quality functions of wetlands resulting from altered fluctuations in water levels.
- Changes in how water levels fluctuate in wetlands have documented impacts on invertebrate and amphibian habitat. Both groups of wildlife exhibit reduced species richness and abundance when wetlands are subject to increased fluctuations in water levels. Impacts to the suitability of wetlands as habitat for mammals, fish, and birds have not been documented.
- Increasing fluctuations in water levels also reduce plant richness in wetlands.

4.6 Impacts of Changing the Amounts of Sediment

4.6.1 Impacts of Changing Amounts of Sediment on Hydrologic Functions

Despite a lack of explicit information on impacts that sedimentation may have on hydrologic functions, it is possible to hypothesize that increases in sediment load to a wetland will reduce the amount of water it can store. For every cubic yard of sediment deposited in a wetland, the storage capacity of water is reduced by a similar amount. This means that wetlands along stream corridors with high inputs of sediment may lose much of their ability to store surface waters during floods. Some wetlands with a lot of erosion in the contributing basin, but not along stream corridors, may also have high inputs.

This is especially true for depressional wetlands. By definition, depressional wetlands function to reduce flooding by storing water behind a restricted outlet and then releasing it slowly. There is less chance in depressional wetlands than in riverine wetlands that severe floods will erode the new sediments and restore the wetland's storage capacity.

4.6.2 Impacts of Changing the Amount of Sediment on Functions that Improve Water Quality

Whether changing the sediment load to a wetland has positive or negative impacts on the water quality functions is not documented in the literature.

4.6.3 Impacts of Changing the Amount of Sediment on Plants

Accelerated sediment deposition or erosion can tax the ability of plant communities to adapt (Kantrud et al. 1989, Jurik et al. 1994, Wang et al. 1994). Sediments have been found to impact plant communities in wetlands in several general ways:

- **Burying seeds, leaves, or plants.** Sedimentation can bury established vegetation and seed banks (Adamus et al. 2001). The burial of leaves prevents photosynthesis and restricts gas exchange through foliage (Ewing 1996). Buried plants expend energy elongating their shoots in an attempt to outpace sedimentation, seeking oxygen and light, and consequently may be less robust.

- **Changing the depth of habitats.** Over the long term, sedimentation can shrink shallow wetlands or reduce the depth of ponds that previously were too deep to support many wetland plants. Such long-term changes in water depth or relative elevation also result in shifts in species composition, as has been documented in the Mississippi River floodplain (Adamus et al. 2001).

- **Inhibiting germination.** Seeds of the most sensitive species often fail to germinate when buried (Dittmar and Neely 1999). The addition of sediment has been found to reduce germination rates of wetland herb species by 34% (Neely and Wiler 1993), 80% (Jurik et al. 1994), and 90% (Wang et al. 1994) depending on the species involved. In general, the species with larger seeds appear to be better able to survive burial (Dittmar and Neely 1999, Jurik et al. 1994, Wang et al. 1994).

  Less than 0.5 inch (1 cm) of sediment can inhibit germination of cattails (*Typha* sp.), barnyard grass (*Echinocola crusgalli*), rice cutgrass (*Leersia oryzoides*), and sedges (*Carex* sp.) (Jurik et al. 1994). Sedimentation inhibits the germination of cattail (*Typha latifolia*) seeds more than seeds of bur-reed (*Sparganium eurycarpum*) (Neely and Wiler 1993). Germination of cattail (*Typha* x *glauca*) seeds decreased by 60 to 90% when sediment loads of less than 0.5 inch (0.2 to 0.4 cm) were applied to the surface of the soil (Wang et al. 1994).

  In contrast, burial by 1 inch (2 cm) of sediment does not interfere with germination of several non-native plant species (Blackshaw 1992, Reddy and Singh 1992).

- **Reducing survival of seedlings.** Excessive sedimentation can reduce the survival of seedlings (Jurik et al. 1994). For example, the density of cattail seedlings and their biomass decreased as sediment loads increased from 0.08 to 0.5 inch (0.2 to 1.0 cm). One study found a fourfold greater density of annuals (vs. perennials) in some heavily sedimented sites (Neely and Wiler 1993). Older and larger seedlings were more tolerant of burial (Wang et al. 1994).

- **Favoring species more tolerant of sediment.** Sedimentation impacts individual wetland species in different ways. The composition of the plant community will therefore change as the most sensitive species are suppressed by the sediments while the more tolerant ones thrive. Effects of sedimentation on particular wetland plant species are not well documented (van der Valk and Jolly 1992) but findings relevant to wetland species found in Washington are discussed here.

  Many mature plants, and especially woody species, apparently are not harmed by a small amount of sedimentation (Wang et al. 1994). Adult plants of wild celery (*Vallisneria americana*) tolerated burial to depths of up to 4 inches (10 cm) but none survived burial under sediment depths of 10 inches (25 cm) (Rybicki and Carter 1986). Among woody plants, saplings of red alder (*Alnus rubra*) tolerated burial less well than those of Oregon ash (*Fraxinus latifolia*) (Ewing 1996).

  Growth of the invasive reed *Phragmites australis,* however, typically keeps pace with moderate rates of sedimentation (Pyke and Havens 1999). However, seeds, seedlings, and plants that have evolved in wetland types in which sedimentation is rare (such as bogs) are highly sensitive to burial. The size of particles that are being deposited, not just their amount, may also influence plant survival (Dittmar and Neely 1999).

4.6.4 Impacts of Changing Amounts of Sediment on Habitat for Invertebrates

In general, increased amounts of sediment can reduce the richness and density of invertebrates and alter their species composition. Excessive sedimentation affects invertebrates through several mechanisms (reviewed in Adamus et al. 2001):

- Burial of detritus and algae, which are important food sources
- Increase in the time required for invertebrates to move through deposited sediment and collect scarce food items
- Reduced flow of water through soil particles, which is necessary to supplying invertebrates with adequate dissolved oxygen
- Mortality of plants that otherwise provide attachment structures and shelter to invertebrates

Some studies have linked changes in invertebrate communities to the development of watersheds (e.g., Hogg and Norris 1991, Ludwa 1994, Carlisle et al. 1998, Ludwa and Richter 2000a). Development often is accompanied by increased export of sediment to water bodies.

Many invertebrate communities in wetlands are adapted to occasional deposition of small amounts of sediment, whereas constant or severe deposition causes major changes. The following bullets summarize some of the studies that have documented impacts of sediment on individual invertebrate species, as well as groups of species, many of which are found in Washington.

- Once deposited, sediments can further damage wetland invertebrate communities if they are resuspended by wind mixing or fish, making the water turbid. For example, bottom-feeding carp (*Cyprinus carpio*) noticeably increase turbidity, both directly (as they move along the bottom) and by consuming aquatic plants that otherwise would stabilize and trap sediments (Lougheed et al. 1998). The biomass of planktonic invertebrates declined in Utah ponds after introduction of carp (Huener and Kadlec 1992).

- In some instances, invertebrate density and perhaps richness can increase over the long term if sedimentation replaces coarser substrates with finer substrates that better support establishment of rooted plants. In temporarily flooded prairie pothole wetlands, only caddisflies seemed relatively unaffected by surrounding land use that generated sediments. Ostracods (seed shrimp), cladocerans (water fleas), and some snails (planorbiids, lymnaeids, physids) were diminished, presumably in part because of sedimentation (Euliss and Mushet 1999).

- Burrowing, tube-forming worms and midges commonly predominate where sediments accumulate (Magee et al. 1993). Filter-feeding species and those that graze on the bottom are most sensitive (Lougheed and Chow-Fraser 1998). However, invertebrate size and behavior also influence their tolerance to sediments (McClelland and Brusven 1980). On the other hand, substrates newly created by sedimentation may attract tolerant individuals and species that are poor competitors on older, more crowded substrates (Soster and McCall 1990).

- Severe and rapid sedimentation is inevitably lethal to nearly all aquatic invertebrates. In North Dakota, wetlands surrounded by cropland were virtually devoid of the resting eggs of zooplankton, whereas such eggs were abundant in wetlands surrounded by mostly natural grassland, which presumably minimized erosion and sedimentation (Euliss and Mushet 1999).

- Unionid mussels (mussels in the family Unionidae) are sensitive to increased sedimentation (Goudreau et al. 1993, Box and Mossa 1999). Numbers of the swamp fingernail clam (*Musculium partumeium*) and amphipods were reduced in willow wetlands in northeastern Missouri where 2 to 4 inches (5 to 10 cm) of sediment had been recently deposited (Magee et al. 1993).

- Sediments may clog the filter feeding mechanisms of some species and limit light penetration. This would adversely impact phytoplankton and other primary producers, with a subsequent adverse impact on food chains (Euliss and Mushet 1999).
- Sedimentation also potentially buries invertebrate eggs deposited in the substrates of wetlands (Euliss and Mushet 1999).

4.6.5 Impacts of Changing Amounts of Sediment on Habitat for Amphibians and Reptiles

Few studies of the impacts of increases in the deposition of sediment on amphibians and reptiles have been conducted in wetlands. On one hand, some species require soft sediments as hibernation sites. For example, painted turtles (*Chrysemys picta*) used sediments 1.6 to 3 feet (0.50 to 0.95 m) thick in an Ontario pond (Taylor and Nol 1989). On the other hand, excessive sediments, when stirred, impair light penetration of the water column and thus can inhibit growth of algae and especially submersed aquatic plants, which provide cover and attachment sites for amphibian eggs.

4.6.6 Impacts of Changing Amounts of Sediment on Habitat for Fish

No recent studies on the impacts of sediment on habitat for fish in North American wetlands or lakes were found. Most of the studies on the impacts of sediment on fish populations have been done in streams, especially as it concerns the growth and reproduction of salmonids in the Pacific Northwest. This information was reviewed and synthesized in Knutson and Naef (1997). The conclusion reached by Knutson and Naef quoted below can also apply to wetlands because streams are often considered a part of wetlands:

> *Sedimentation in fish-bearing waters affects habitat quality and fish survival in a number of ways. Stream bottoms covered with fine sediments are no longer suitable for spawning. Sediments cover and suffocate fish eggs and fry. High sediment deposits also block fish passage to upper spawning reaches. Suspended sediments clog the gills of fish, decrease dissolved oxygen levels, inhibit fish feeding and growth, and suppress macroinvertebrate food sources.*

4.6.7 Impacts of Changing Amounts of Sediment on Habitat for Birds

Little information was found on how sedimentation impacts the habitat that a wetland provides for bird communities. One can hypothesize, however, that sedimentation will impact birds by altering habitat structure, killing submersed vegetation, or altering the

abundance or availability of prey items. In one case, however, densities of breeding dabbling ducks were positively correlated with wetland turbidity in ponds in the interior of British Columbia (Savard et al. 1994).

4.6.8 Impacts of Changing Amounts of Sediment on Habitat for Mammals

How sedimentation impacts the habitat that a wetland provides for mammals was not documented in the literature. As with birds, however, one can hypothesize that sedimentation can impact mammals by altering habitat structure or changing the abundance or availability of prey items.

4.6.9 Summary of Key Points

- Impacts to the hydrologic functions from sedimentation can be hypothesized because an increase in sediments reduces the storage capacity of a wetland.
- No information was found on possible impacts of sedimentation on the functions of wetlands that improve water quality.
- Increasing sedimentation will also decrease plant richness and tends to favor the more invasive types that tolerate disturbance.
- Impacts of increased amounts of sediment on the habitat functions of wetlands have been documented for invertebrates, amphibians, and fish. All of these groups generally have reduced species richness and abundance when wetlands are subject to increased sedimentation. In some cases, however, where the sediments coming into a wetland are finer than existing sediments, the number of invertebrate species may increase. Impacts from sedimentation on the suitability of wetlands as habitat for mammals and birds have not been documented.

4.7 Impacts of Increasing the Amount of Nutrients

The major nutrients for plant growth, phosphates, nitrates, and ammonium, can be transported into aquatic systems in dissolved forms or adsorbed onto sediment. The element phosphorus is usually the limiting nutrient for plant growth in freshwater aquatic systems (Newton 1989, Mitsch and Gosselink 2000). Because it is the limiting factor, phosphorus in the presence of the other critical element, nitrogen, allows expansive growth of phytoplankton, algae, and larger plants in aquatic systems when it is available in higher quantities.

Excessive algal growth is unsustainable, and when the algae blooms die, their decomposition causes the available dissolved oxygen to be consumed. This cycle of

excessive plant growth, plant death, and decomposition that uses up oxygen is called *eutrophication*.

Excess phosphorous and nitrogen, therefore, often leads to eutrophication with subsequent mortality of the aquatic organisms that require oxygen (Newton 1989, Mitsch and Gosselink 2000). Wetlands with areas of water on the surface can therefore become eutrophic if they receive excessive amounts of phosphorus and/or nitrogen.

4.7.1 Impacts of Increasing Amounts of Nutrients on Hydrologic Functions

It is possible that the stimulation of plant growth by excess nutrients could increase the density of plants in the wetland. A thicker stand of vegetation can be expected to provide more resistance to flood flows than a thinner one (Hruby et al. 1999). Therefore, excess nutrients might indirectly improve the reduction in velocity that a wetland provides during floods. The literature did not provide any other information on how nutrient impacts might affect the hydrologic function of wetlands.

4.7.2 Impacts of Increasing Amounts of Nutrients on Functions that Improve Water Quality

Some research indicates that excessive nutrients from agricultural operations may reduce the normal ability of wetland microbes to detoxify particular pesticides (Kazumi and Capone 1995, Chung et al. 1996, Entry and Emmingham 1996). Adding nitrogen to riparian wetlands may potentially compromise the long-term ability of the system to remove nitrogen via denitrification (Ettema et al. 1998). Other information on this topic was not documented in the literature.

However, several avenues of research could be combined to make some hypotheses about impacts. The addition of nutrients to acidic bogs results in changes in plant communities. The plant community that maintains the high acidity in the bog may change to one that maintains a more neutral pH. These changes might then alter several aspects of chemistry in the wetland that affect its ability to improve water quality. The rate of nitrification will probably increase because, as noted by Mitch and Gosselink (2000), low pH inhibits denitrifying bacteria. The change in pH will also probably change the ability of the wetland to bind different toxic metals and other compounds. (See the discussion in Chapter 2 on how pH is linked to the ability of a wetland to bind different pollutants.)

4.7.3 Impacts of Increasing Amounts of Nutrients on Plants

Excessive nutrients can affect wetland plants in a variety of ways including:

- Shifting the species composition away from species that take up nutrients slowly, to those that are able to exploit nutrient pulses more rapidly or which have high

nutrient requirements (Hough et al. 1989, Arts et al. 1990, Gopal and Chamanlal 1991, Wetzel and van der Valk 1998)

- Triggering algal blooms that can shade out many submersed herbaceous plants (Crowder and Painter 1991, Stevenson et al. 1993, Srivastava et al. 1995, Short and Burdick 1995)
- Causing dead plant material to accumulate faster than it can decompose completely, thus altering understory and soil structure (Neill 1990b, Craft and Richardson 1993)

Such changes usually result in long-term changes in the distribution and richness of plants within the wetland. Over the long term, nutrient additions to most wetlands tend to reduce species richness and increase the dominance of a few species. Often, non-native species are most capable of invading rapidly changing environments. Consequently they frequently come to dominate some nutrient-enriched wetlands (Adamus et al. 2001).

Increases in plant litter can smother other plants when the fast growing species die, thus helping maintain the dominance of species that exploit nutrients the most (Adamus et al. 2001). For example, the addition of nitrogen and phosphorus fertilizers to a marsh dominated by cattail (*Typha glauca)* and the grass *Scolochloa festucacea* during two growing seasons resulted in increased biomass of both species. However, the biomass of *S. festucacea* declined in the second year due to accumulated litter of *T. glauca* (Neill 1990b).

The plants in bogs and other nutrient-poor wetlands are logically the most sensitive to nutrient additions (Moore et al. 1989). The increased availability of nutrients allows grasses and common opportunistic plants to outcompete rare plants (such as sundews, orchids, and pitcher plants) that are adapted to nutrient-poor conditions. For example, in Appalachian peat bogs, the spatial dominance of bristly dewberry (*Rubus hispidus*) was positively related to nutrient levels, but dominance of the Ericaceae shrubs was negatively related (Stewart and Nilsen 1993).

Many aquatic plant species respond to nutrient additions with increased growth, biomass, and productivity. Growth responses to enrichment have been documented for about 80 wetland-associated species in North America. Of these, most have tolerated enrichment or responded to enrichment with increased biomass or growth (Adamus and Gonyaw 2000).

Information on the response of many individual plant species to nutrients can be found in the National Database of Wetland Plant Tolerances at: http://www.epa.gov/owow/wetlands/bawwg/publicat.html#database1

4.7.4 Impacts of Increasing Amounts of Nutrients on Habitat for Invertebrates

Excessive nutrients can cause long-term and short-term shifts in invertebrate communities. The information available suggests that excess nutrients can result in both decreases and increases in species richness as well as changes in the groups of invertebrates found. The direction of the change depends on how the nutrients impact the vegetation and soils that are the main habitat for invertebrates. Findings from the literature include:

- **Increased richness of invertebrates.** Up to some point, nutrient inputs to wetlands can lead to increased invertebrate richness, as more food sources become available to predatory invertebrates (Rader and Richardson 1992, Campeau et al. 1994, Cieminski and Flake 1995, Gernes and Helgen 1999).

- **Reduced richness of invertebrates.** Invertebrate richness in a series of highly enriched wastewater wetlands was found to be lower than in a less enriched reference wetland (Nelson et al. 2000).

- **Changes in the types of invertebrates.** In some cases excess nutrients result in the increased dominance of certain kinds of algae. Invertebrates that specialize in feeding on these algae, or that characteristically find shelter and attachment sites in the aquatic plants, then have an advantage and can become dominant (Murkin et al. 1991, Campeau et al. 1994). Exposure to organic enrichment and eutrophication frequently causes an increase in grazers (such as Tanypodinae midges), as well as other herbivores, species that feed on detritus, predators, and "miners" that burrow into plants. These are groups that typically increase with increasing growth of algae growing on the bottom and emergent aquatic plants (Campeau et al. 1994). A study of four lacustrine/bay wetlands bordering Lake Michigan also found that midge communities shifted across nutrient gradients (Murkin et al. 1994, Campeau et al. 1994).

- **Increased density of invertebrates.** Total invertebrate density increases with increased nutrients, as algal production becomes less of a limiting factor in the invertebrate community (Murkin et al. 1992, Campeau et al. 1994).

- **Changes in the bioaccumulation of metals by invertebrates.** Nutrients appear to influence the tendency of aquatic invertebrates to accumulate heavy metals and the type of metals that are accumulated. For instance, zinc, iron, and manganese concentrations were higher in midges from nutrient-rich wetlands, whereas high copper concentrations were found in midges from nutrient-poor wetlands (Bendell-Young et al. 1994). This may be due at least partly to the bioavailability of various metals being influenced by oxygen conditions in the sediment, which in turn are partly the result of decomposition of algal blooms triggered by high concentrations of nutrients (Adamus et al. 2001).

4.7.5 Impacts of Increasing Amounts of Nutrients on Habitat for Amphibians and Reptiles

The review of the literature indicates that amphibians can be impacted by the input of nutrients. No studies were found on impacts on reptiles.

Amphibians in the Northwest can be directly impacted by the input of nitrates. Five amphibian species in Oregon showed both sublethal responses and mortality following laboratory applications of nitrate. These studies indicated that the EPA nitrate criteria for drinking water of 10 milligrams per liter (mg/l) and/or for protection of warmwater fish are inadequate to protect these amphibians (Marco et al. 1999). In Texas, playa wetlands receiving nutrient-laden effluent from feedlots were devoid of amphibians found in natural playas (Chavez et al. 1999). Experiments indicated that effluent had to be diluted to less than 3% strength in order to minimize adverse effects on the leopard frog (*Rana pipiens*).

Indirect impacts of excessive nutrients can also be important to amphibians. Shifts in seasonal timing and amount of nutrients that enter a wetland can, over a period of years, increase the relative dominance of algae and/or emergent plants at the expense of submersed plants. This in turn can reduce the availability of submersed plants as attachment substrates for amphibian eggs and as cover for larvae (Beebee 1996).

Excess nutrients can also diminish dissolved oxygen levels (Tattersall and Boutilier 1999), alter the abundance of aquatic predators, and shift the algal and invertebrate foods available to amphibians (Horne and Dunson 1995). As a result, species composition and sometimes species richness of amphibian communities can decline as eutrophication becomes severe. However, well designed studies of such effects are few.

4.7.6 Impacts of Increasing Amounts of Nutrients on Habitat for Fish

Direct impacts of excess nutrients on fish in wetlands were not documented in the literature. However, the secondary impacts of eutrophication such as oxygen depletion do affect fish. Much of the literature deals with impacts of low oxygen in streams (for a review see Knutson and Naef 1997), and it can be assumed that the impacts of low oxygen in wetlands will be similar.

As mentioned previously, the increased plant production that results from added nutrients often results in low oxygen levels when the plant material dies and starts to decompose. Many fish species suffer from reduced levels of dissolved oxygen, and feeding habits also may shift. To some degree, fish families can be grouped according to their susceptibility to oxygen deficiencies. Salmonids and coregonids (whitefish) require high levels of dissolved oxygen, whereas cyprinids (a large family that includes carp and goldfish) often tolerate low dissolved oxygen levels (Harper 1992). Thus the species composition and richness may change depending on the initial state of the wetland and the duration and magnitude of the eutrophication.

4.7.7 Impacts of Increasing Amounts of Nutrients on Habitat for Birds

Eutrophication can indirectly impact the composition of the wetland bird community by altering the vegetation structure and availability of prey. In general, moderately elevated nutrient levels also spur the growth of submersed plants that provide food for ducks, as well as supporting more aquatic insects that are especially important as food for ducklings and aerial foragers like swallows. However, excessive nutrients cause algal blooms that can kill fish eaten by birds, reduce the growth of plants growing on the bottom by blocking light, and reduce visibility of other food items under the water surface.

Studies that have documented changes in the bird community related to excess nutrients are summarized below:

- Excessive nitrates have been implicated in deaths of some frogs (see Section 4.7.5). Frogs are significant prey item for many wetland birds (Adamus et al. 2001).
- Northern shoveler (*Anas clypeata*) and eared grebe (*Podiceps nigricollis*) were positively associated with phosphorus in a survey of wetlands in interior British Columbia (Savard et al. 1994).
- Water-bird abundance and biomass were positively correlated in 46 Florida lakes with levels of phosphorus, nitrogen, and chlorophyll. There also was a positive correlation of water-bird richness with phosphorus, after accounting for nutrients contributed to the lakes by the birds themselves (Hoyer and Canfield 1994).
- Total density of dabbling ducks was correlated positively with total dissolved nitrogen (Savard et al. 1994).
- The parasitic nematode *Eustrongylides ignotus,* which has only been found in disturbed and enriched wetlands (Spaulding and Forester 1993), negatively affects the health of adult wading birds and the survival of their nestlings (Spaulding et al. 1993).

4.7.8 Impacts of Increasing Amounts of Nutrients on Habitat for Mammals

Impacts from increases in nutrients on the habitat of mammals in wetlands are not described in the literature. It can be hypothesized, however, that if eutrophication results in anoxic conditions that are lethal to the prey of mammals (e.g., fish and some amphibians), the community composition may shift from predator species (such as otter or mink) to vegetarian or invertebrate-eating species and opportunists (such as muskrat).

4.7.9 Summary of Key Points

- Some impacts to the hydrologic functions from increased nutrients can be hypothesized because the increased growth of plants resulting from increased nutrients may provide better resistance to the movement of flood waters.
- Some impacts to the functions of improving water quality have been reported. These include a potential reduction in the ability of wetlands to detoxify pesticides and to remove nitrogen as a pollutant. Impacts from increased nutrients can also be hypothesized for bogs. The ability of bogs to bind toxic metals may be reduced but their ability to remove nitrogen may be increased.
- Increasing nutrients will stimulate plant growth and may change the composition of the species present.
- Impacts of increased amounts of nutrients on the habitat wetlands provide have been documented for invertebrates, amphibians, and birds. Excess nutrients can result in both an improvement in the habitat through the production of food and a reduction in habitat through eutrophication. The actual impacts depend on local conditions in the wetland. Impacts to the habitat for fish and mammals can be inferred because eutrophication causes reductions in the levels of oxygen in the water with resultant impacts to both water quality and the food sources for these two groups.

4.8 Impacts of Increasing the Amount of Toxic Contaminants

4.8.1 Impacts of Increasing Amounts of Toxic Contaminants on Hydrologic Functions

No explicit information was found in the literature on the possible impacts of toxicity from contaminants on the hydrologic functions provided by wetlands (storing flood waters, reducing erosion, and recharging groundwater).

4.8.2 Impacts of Increasing Amounts of Toxic Contaminants on Functions that Improve Water Quality

Information on how toxic compounds affect the function of wetlands to remove pollutants is sparse. It can be hypothesized, however, that an input of low levels of toxic compounds may stimulate the ability of a wetland to detoxify pollutants. Some microbial species biodegrade particular contaminants and their abundance is increased in the presence of low levels of the contaminants. These species can flourish in some wetlands that are only mildly or moderately contaminated.

Contaminants that can be processed by microbes when at low to moderate concentrations include copper (Farago and Mehra 1993), mercury (Marvin-Dipasquale and Oremland 1998), selenium (Steinberg and Oremland 1990, Azaizah et al. 1997), cadmium (Sharma et al. 2000), manganese (Sikora et al. 2000), and petroleum (Nyman 1999, Megharaj et al. 2000).

4.8.3 Impacts of Increasing Amounts of Toxic Contaminants on Plants

Most plant species are relatively tolerant to toxic contaminants. Impacts usually result from the effects of contaminants on plant metabolic pathways, enzymatic reactions, and growth (Fitter and Hay 1987). Symptoms of toxicity can include reduced growth; small, discolored, or dying leaves; early leaf fall; and stunted or suppressed growth of roots (Pahlsson 1989, Rhoads et al. 1989, Vasquez et al. 1989).

Shifts in the composition of the plant community in response to contaminants have not been widely documented. Relevant studies include:

- Arsenic, cadmium, copper, lead, and zinc inhibited growth in hybrid poplar (*Populus*) and several other tree species (Lejeune et al. 1996).
- Iron and manganese, although not usually toxic to wetland plants, do affect species in some wetland types. For example, laboratory experiments revealed differences among 44 fen species with regard to the influence of iron on growth (Snowden and Wheeler 1993).
- Oil spills can have long-lasting effects on wetland plant communities (Obot et al. 1992). In a greenhouse experiment, oil and a detergent used to clean up oil spills were applied to broadleaf arrowhead (*Sagittaria lancifolia)*, salt marsh sedge (*Scirpus olneyi),* and common cattail (*Typha latifolia)*. The leaves on all of the study plants died following oiling, but new leaves soon developed on those plants subjected to oil and subsequent cleaning with the detergent. *S. olneyi* was the least sensitive of the three species, whereas *T. latifolia* appeared to be the most sensitive (Pezeshki et al. 1998).
- The herbicides Rodeo® and Garlan 3A®, applied to control purple loosestrife (*Lythrum salicaria*), also reduced the growth rates of non-target species such as duckweed (*Lemna gibba)* (Gardner and Grue 1996).

4.8.4 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Invertebrates

General studies on the impacts to invertebrates in wetlands of Puget Sound found that increased levels of toxic contaminants and changes in the water regime resulted in declines in taxa richness among the scraper and shredder functional feeding groups and

the Chironomidae family (small, mosquito-like flies) (Ludwa and Richter 2001). These authors found declines in richness and abundance of invertebrate groups whose presence is seen as an indicator of the general health or quality of a water body. A second study in the Pacific Northwest also showed a direct and negative correlation between urbanization and the abundance and richness of macroinvertebrates (Hicks 1995) primarily through impacts to water quality.

The following sections first review the effects of metals on invertebrates and then describe the effects of organic and synthetic compounds such as pesticides. Much of the information on the impacts on invertebrates is based on studies in streams. These studies are applicable to wetlands because the species and groups of species reported in the studies are also found in wetlands.

4.8.4.1 Effects of Heavy Metals on Invertebrates

Heavy metals such as mercury, lead, zinc, copper, and cadmium can be directly toxic to wetland invertebrates. Metals can also impact invertebrate communities by altering the species composition and abundance of algae and aquatic plants upon which invertebrates depend for food and shelter. Growth, larval development, and reproduction of invertebrates can also be harmed by long-term exposure to sublethal concentrations of trace metals (Timmermans 1993). Relatively little, however, is known about the sublethal effects of metal pollutants in freshwater wetlands or how metals are metabolized or accumulated.

The extent to which heavy metals are toxic to wetland invertebrates depends largely on the acidity of the wetland and the particular form of the metal involved. Acidic conditions can mobilize and increase the toxicity of some metals, such as cadmium (Wright and Welbourn 1994), and decrease the toxicity of others, such as aluminum (Wren and Stephenson 1991). On the other hand, some metals, such as iron and aluminum, can to some degree protect invertebrates from otherwise toxic effects of heavy metals in acid mine drainage (Whipple and Dunson 1992).

Specific studies documenting the impact of heavy metals on invertebrates are summarized below:

- More than 20 years after cadmium and cobalt discharges to a freshwater marsh in New York were curtailed, invertebrate richness remained lower than at a control (less polluted) site (Klerks and Levinton 1993).
- Moderate recovery of invertebrates from metal contamination was demonstrated in the Coeur D'Alene River in Idaho. Over 22 years after contamination by zinc and other metals ceased, the number of taxa grew from zero to 18, while the proportion of mayflies, stoneflies, and caddisflies relative to the proportion of midges rose (Hoiland and Rabe 1992, Hoiland et al. 1994).

- Some studies show herbivores and detritivores as the most sensitive to additions of metals (Kiffney and Clements 1994a, Leland et al. 1989), whereas others have reported scrapers being the most sensitive group (Clements 1994).
- Mayflies and some stoneflies of western streams are sensitive to metals, whereas caddisflies and midges are relatively tolerant (Clements 1994, Kiffney and Clements 1994b, Leland et al. 1989, Nelson and Roline 1996).
- Agricultural drainage water containing arsenic, boron, lithium, and molybdenum entering the Stillwater Wildlife Management Area in Nevada proved acutely toxic to many wetland invertebrates (Hoffman 1992, Hallock and Hallock 1993a, 1993b).
- Copper and some other heavy metals appear to be more damaging to aquatic communities in the spring and summer rather than in the fall (Leland et al. 1989). Summer exposure to metals may coincide more closely with hatching of many macroinvertebrates, and early periods in the development of the invertebrates may be more susceptible.

4.8.4.2 Effects of Pesticides, Oil, and Other Contaminants on Invertebrates

Pesticides, oil, and other toxic contaminants represent a wide range of pollutants. In general, however, most have been shown to change the community structure (abundance, distribution, and richness) of invertebrates. Contaminants cause these effects through several mechanisms, including:

- Causing acute or chronic toxicity to invertebrates
- Altering algal communities and aquatic plants upon which some invertebrates depend for food and shelter
- Altering predation on invertebrates by decimating numbers of other crustaceans, fish, and amphibians
- Reducing rates of oxygen diffusion
- Changing the effects of other potential disturbances, such as acidity

The range of pesticides and organic pollutants used today is very large and it is not possible to generalize the impacts of this group of pollutants on invertebrates. Table 4-1 summarizes numerous studies that demonstrate the wide range of responses to contaminants by invertebrates.

Table 4-1. Summary of studies on effects of contaminants on invertebrates.

Reference	Contaminant Studied	Results
Eisler (1992)	diflubenzuron (insecticide)	In laboratory tests diflubenzuron was most toxic to crustaceans, followed by mayflies, midges, caddisflies. Larvae of corixids, dragonfly adults and larvae, spiders, dytiscids, and ostracods had moderate sensitivity
Eisler (1992)	paraquat, cyanide, fenvalerate, acrolein	These substances were lethal to invertebrates
Dieter et al. (1996)	phorate (pesticide)	In Prairie Pothole Region, macroinvertebrates that were particularly sensitive to phorate included hemipterans, mosquitoes, flies, mayflies, water mites, and water beetles. Less sensitive were leeches, snails, aquatic worms, ostracods
Lieffers (1990)	3-trifluoromethyl-4-nitrophenol (TFM) (lampricide)	TFM had a significant effect on invertebrates in a small stream
Fairchild and Eidt (1993)	fenithrothion (insecticide for forest insects)	Fenithrothion reduced emergence of aquatic insects for 6 to 12 weeks. Densities of most invertebrates (especially predatory species, midges, some other dipterans) were reduced by as much as 50% for more than one month after treatment. Wetland sediments became dominated by aquatic worms and water mites. Although in many streams and large lakes fenithrothion has transitory effects, residual toxicity remained in bog wetlands during winter and into the next year
Hachmoller et al. (1991)	various organic pollutants	Mayflies, stoneflies, caddisflies decreased in abundance in stream contaminated by various organic pollutants
Keller (1993), Metcalfe and Chaarlton (1990)	various contaminants	Mussels are especially sensitive to combined effects of pesticides, organic compounds, excessive nutrients
Kemp and Spotila (1996)	industrial pollutants, PCBs	Isopods, oligochaetes, craneflies were main survivors in a Pennsylvania stream with industrial pollution (including PCBs) compared with non-urbanized control segments
Crunkilton and Duchrow (1990)	oil	After 25 days, an oil spill in a Missouri stream reduced macroinvertebrate population to less than 0.1% of normal densities. Recovery of some species of stoneflies, mayflies, and caddisflies did not occur for at least nine months
Henry et al. (1994)	surfactant	In laboratory tests, a surfactant was approximately 100 times more toxic than the herbicide glysophate, with which it is commonly applied
Wipfli and Merritt (1994), Kreutzweiser et al. (1994a), Jackson et al. (1994), Waalwijk et al. (1992)	*Bacillus thuringiensis* var. *israelensis* (Bti) (biological control agent)	Bti appears to have minimal adverse effects on non-target insects in streams although mortality has been observed in Lepidoptera, some midges, craneflies, caddisflies, mayflies

Reference	Contaminant Studied	Results
Euliss and Mushet (1999)	agricultural contaminants	Direct adverse correlation found between aquatic invertebrate species richness and agricultural practices for seasonally inundated wetlands in prairie pothole region of North Dakota. Adverse effects on invertebrates could result from agrichemicals (shown to cause increased mortality of aquatic invertebrates in other studies). Tilling around wetland could increase erosion, leading to suspended sediments and adsorbed metals that are toxic to some zooplankton and thus affect the food chain

4.8.5 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Amphibians and Reptiles

Studies of the effects of heavy metals, pesticides, and other toxins on amphibians and reptiles have been conducted mainly on species, not communities. A review of relevant literature was published by Sparling et al. (2000). Schuytema and Nebeker (1996) have compiled a database of toxicity information from published literature for 58 amphibian species as related to 135 chemicals.

Many different pollutants have been documented as toxic to species of amphibians and reptiles found in Washington's wetlands. The following references document the impact of toxic compounds on some species found in the Pacific Northwest:

- Toxic effects of aluminum and other metals on the embryos and tadpoles of the northern leopard frog (*Rana pipiens*) were found by Freda (1989, 1991), Freda and McDonald (1990), and Freda et al. (1990).
- Many synthetic organic compounds affect amphibians and aquatic reptiles. Northwestern salamander (*Ambystoma gracile*) egg mortality corresponded with levels of total petroleum hydrocarbons in western Washington (Platin 1994, Platin and Richter 1995).
- The pesticide esfenvalerate caused damaging sublethal effects on tadpoles of the northern leopard frog (Materna et al. 1995).
- Tests of three forest insecticides (fenitrothion, triclopyr, and hexazinone) on the northern leopard frog in Ontario suggested that tadpoles were sensitive to triclopyr and fenitrothion (Berrill et al. 1991).

4.8.6 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Fish

The response of fish communities and individual species to toxic compounds is varied and complicated by many environmental factors. Smaller fish may be the first to respond to contaminants (Matuszek et al. 1990).

The toxicity of copper and zinc to some fish species depends on other chemical characteristics of the water (Munkittrick and Dixon 1992, Welsh et al. 1993, Erickson et al. 1996), as well as fish behavior (Pourang 1995). For example, dissolved organic matter from a marsh at a level of 5 mg carbon per liter kept copper from binding to the gills of small steelhead (*Oncorhynchus mykiss*), thereby reducing its toxicity. This occurred because copper formed a complex with dissolved organic carbon, making the copper unavailable (Hollis et al. 1997). In addition, some fish species may acclimate to moderately elevated levels of some metals (Klerks and Lentz 1998).

Selenium is not directly toxic to fish at usual concentrations but can become toxic once concentrated in fish food chains. This is especially true in some wetlands that receive effluents from irrigated fields or power plant reservoirs in some regions (Zilberman 1991, Lemly 1996).

Synthetic organics, including pesticides, can accumulate in wetland fish (Cooper 1991), often with adverse effects. In a Canadian wetland receiving effluent containing oily sand, fish had altered blood chemistry and died within 14 days (Bendell-Young et al. 2000).

4.8.7 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Birds

The response of individual bird species and bird communities to toxic compounds is varied. Individual species are directly affected by many pollutants. Many pesticides, however, are more likely to impact bird populations by altering their habitat and foods rather than by direct toxicity. Studies that document such impacts are summarized below:

- Several instances have been documented of wetland birds being directly poisoned by insecticides applied at recommended rates (e.g., parathion, as documented by Flickinger et al. 1991).
- Herbicides have been applied to wetlands to change the structure of vegetation and the species composition, with consequent shifts in the composition of bird species (Solberg and Higgins 1993, Linz et al. 1996). Information on pesticides in prairie wetlands has been compiled by Facemire (1992).
- Evidence of bird toxicity from lead shot used in hunting has been reported by Havera et al. (1992), Hohman et al. (1993), and Peters and Afton (1993).
- Detrimental reproductive effects from dioxins have been documented for great blue herons (*Ardea herodias*) (Hart et al. 1991); for dioxins and furans on wood ducks (*Aix sponsa*) (White and Seginack 1994); for PCBs (polychlorinated biphenyls) in American kestrels (*Falco sparverius*); and for petroleum in mallards (*Anas platyrhynchos*) (Holmes and Cavannaugh 1990).
- Research has continued to focus on the effects of selenium on waterfowl in western states. Biogeochemical conditions favoring the release of selenium into

wetlands are found throughout the arid regions of the western states and threaten bird communities in many wetlands along the Pacific and Central Flyways (Paveglio et al. 1992). Agricultural drainage, irrigation, and natural waters can leach selenium from many western soils. Subsurface irrigation is the most widespread and biologically important source of selenium toxicity for waterfowl, including the waterfowl in six national refuges (Ohlendorf et al. 1990, Feltz et al. 1991). Selenium is often accompanied by boron, which is toxic to ducklings (Stanley et al. 1996).

Impacts of lead shot

Lead in the aquatic environment can have significant impacts. Lead is toxic to aquatic biota (Eisler 1988). Waterborne lead is the most toxic form. Waterborne concentrations over 10 micrograms per liter have significant long-term effects on fish (Demayo et al. 1982). The introduction of lead into the aquatic food chain via aquatic plants has been found in the roots and foliage of the pond weed *Potamogeton foliosus* and in the exoskeleton of crayfish (Eisler 1988, Knowlton et al. 1983). Elemental lead (lead shot) has been shown to be significantly less bioavailable to rooted aquatics than powdered lead (Behan et al. 1979).

Waterfowl are at risk from ingesting lead shot as they forage in wetlands. Because of the proximity of wetlands to shooting ranges, other aquatic organisms, including amphibians, and some bird species may be at risk from the spent lead. For example, Eisler (1988) found that lead in tadpoles might contribute to the lead levels reported in wildlife that eats tadpoles. Predatory animals that feed on amphibians include reptiles (such as the garter snake), birds such as the great blue heron and red tailed-hawk, and mammals such as raccoons and coyotes (Meehan Martin, personal communication). The cleared range areas also encourage the introduction of rodent populations, which are preyed upon by the same predatory animals listed above.

Herbivorous land snails have been found to play an important role in cycling of lead in contaminated ecosystems (Dallinger and Wieser 1984, Beeby 1985). Therefore, snails and slugs in the forest ecosystem that graze in the gun range area may cycle lead into the forest food chain.

Plants growing in soils of low pH or low organic content readily accumulate lead (Demayo et al. 1982). Application of lime or phosphate, however, converts lead to hydroxides, carbonates, or phosphates of low solubility and reduces uptake by plants (Demayo et al. 1982). This in turn would reduce the amount of lead introduced into the food chain by herbivores.

4.8.8 Impacts of Increasing Amounts of Toxic Contaminants on Habitat for Mammals

Possible impacts of toxicity from pollutants on wetland mammals were not documented in the literature.

4.8.9 Summary of Key Points

- No information was found on the impacts of contaminants on the hydrologic functions of wetlands.
- The rates at which wetlands remove toxic compounds may actually be improved under low levels of contamination because the specific microbes that detoxify the pollutants are stimulated.
- The impact of contaminants on plants has not been studied as extensively, but the information suggests that toxicity from contaminants can change the composition of the plant community.
- Impacts of increased contaminants on the habitat provided by wetlands have been documented for invertebrates, amphibians, fish, and birds. Many contaminants are toxic to these species and their presence in wetlands reduces the suitability of a wetland as habitat. Mammals are the only group of vertebrates for which no information exists in wetlands.

4.9 Impacts of Changing Acidity

4.9.1 Impacts of Changing Acidity on Hydrologic Functions

No information was found on the impacts that increasing acidity might have on the hydrologic functions performed by wetlands. In the absence of any information to the contrary, however, it is possible to hypothesize that decreasing pH will probably not change how wetlands perform these functions. Changes in the acidity of water are not expected to change how well wetlands store water, how well they slow it down during peak flows, or how well they recharge groundwater.

4.9.2 Impacts of Changing Acidity on Functions that Improve Water Quality

Increased acidity (reduced pH) could change aspects of wetland chemistry that affect the ability to improve water quality. The rate of nitrification will probably decrease because, as noted by Mitch and Gosselink (2000), low pH inhibits denitrifying bacteria. The

change in pH will also probably change the ability of the wetland to bind different toxic metals and other compounds.

No other information was found on the impacts that increasing acidity might have on how well wetlands remove pollutants.

4.9.3 Impacts of Changing Acidity on Plants

The pH is critical in determining the distribution of plants in wetlands. Changes in pH that result from human activities can, therefore, have major impacts. Studies described below have documented changes in plant populations that resulted from both decreases in pH (more acidic conditions) and increases in pH (less acidic conditions). However, the effects of acidification (or its reversal by liming) on the species composition of plants are not consistent among wetland types or even within individual wetlands (Farmer 1990, Baker and Christensen 1990, Mackun et al. 1994, Weiher et al. 1994).

For example, many plant species that inhabit bogs are adapted to acidity levels that would kill most wetland plants. Species whose decline or disappearance from a lacustrine wetland coincided with acidification include water lobelia (*Lobelia dortmanna),* shore quillwort (*Isoetes riparia*), water milfoil (*Myriophyllum tenellum)*, yellow pond lily (*Nuphar* sp.), common bladderwort (*Utricularia vulgaris)*, and ribbon leaf pondweed (*Potamogeton epihydris)* (Farmer 1990). Species whose relative abundance increased included *Leptodictium riparium,* needle spike rush (*Eleocharis acicularis),* sphagnum moss *(Sphagnum* sp.), and pipe wort (*Eriocaulon septangulare)* (Farmer 1990).

In general, making wetlands more acidic can directly impact plants by limiting the availability of some inorganic nutrients and carbon (Farmer 1990). Acidic conditions also promote the conversion of nitrates into ammonium.

Acidic conditions can impact plants indirectly by reducing the densities of invertebrates that graze or process detritus. Acidic conditions in wetland soils increase the toxicity of aluminum and manganese (Rendig and Taylor 1989, Crowder and Painter 1991).

4.9.4 Impacts of Changing Acidity on Habitat for Invertebrates

In general, changing the acidity in a wetland can alter the community structure of invertebrates by:

- Causing acute or chronic damage to tissues of invertebrates; species that easily lose sodium ions when pH is reduced tend to be most sensitive (Steinberg and Wright 1992)
- Altering algal communities and aquatic plants upon which some invertebrates depend for food and shelter (see discussion in Section 4.9.3)

- Altering the populations that are predators of invertebrates such as other crustaceans, amphibian, and fish (see Sections 4.9.5, 4.9.6)

The impacts of acidification on aquatic invertebrate communities have been researched extensively. Much of the information from Europe is compiled by Johnson et al. (1993). Table 4-2 categorizes invertebrate species as more or less tolerant of acidification based mainly on the North American literature. The list is included here because many of these species are probably found in Washington's wetlands. Few local studies, however, document the distribution of invertebrates in the state so it is not possible to identify the tolerance of species that are found here.

Some invertebrates are sensitive to pH increases (decreased acidity). For example, stormwater input to a Florida freshwater marsh increased phosphorus levels, lowered oxygen levels, and raised pH and hardness. This resulted in a shift of the macroinvertebrate population toward species that otherwise are intolerant of the acidic, nutrient-poor conditions typically found in the studied wetland (Graves et al. 1998).

Acidity often reduces the richness of macroinvertebrates in aquatic habitats (Schell and Kerekes 1989, Hall 1994). Another study showed that with increased acidity, many aquatic invertebrates declined in numbers and biomass, especially in wetlands with pH below 5.0 (Parker and Wright 1992). Reductions in acid emissions from some Canadian smelters was followed by significant increases in richness of invertebrates in water bodies downwind of the smelters (Griffiths and Keller 1992).

Table 4-2. Summary of studies describing relative tolerance of invertebrates to acidification.

Taxonomic Group and Study Reference	More Tolerant (Less Sensitive)	Less Tolerant (More Sensitive)
Dragonflies and Damselflies (Odonata)		
Damselflies (Parker and Wright 1992, Baker and Christensen 1990)	X	
Some Odonata (*Enallagma civile*) (Giberson and MacKay 1991)		X
Beetles (Coleoptera)		
Some water beetles (Parker and Wright 1992), especially hydrophilid and dystiscid beetles (Baker and Christensen 1990)	X	
True Bugs (Hemiptera, Homoptera)		
Some water bugs, at least Notonectidae, Gerridae, Corixidae (Baker and Christensen 1990)	X	
Some water bugs (Parker and Wright. 1992)		X
Caddisflies (Trichoptera)		
Some caddisflies: *Cheumatopsyche pettiti* (Camargo and Ward 1992).	X	

Taxonomic Group and Study Reference	More Tolerant (Less Sensitive)	Less Tolerant (More Sensitive)
Some caddisflies (Parker and Wright 1992) and some in the scraper and predator guilds (Williams 1991)		X
Flies, Midges, Mosquitoes (Diptera)		
Midges (Havens 1994a, Baker and Christensen 1990, Tuchman 1993), blackflies (Baker and Christensen 1989)	X	
Some midges, such as *Tanytarsus, Microtendipes*, and *Nilothauma* (Griffiths 1992)		X
Stoneflies (Plecoptera)		
Some stoneflies (Tuchman 1993) such as *Amphinemura* and *Leuctra* (Griffith et al. 1995)	X	
Many stoneflies, e.g., *Peltoperla arcuata* (Griffith et al. 1995)		X
Mayflies (Ephemeroptera)		
The mayfly *Eurylophella funeralis* (Griffith et al. 1995)	X	
Some mayflies (Balding 1992)		X
Other Macroinvertebrates		
Planarian *Dugesia dorotocephala* (Camargo and Ward 1992)		X
Some water mites (Havens 1994a)	X	
Molluscs (Grapentine and Rosenberg 1992, Gibbons and Mackie 1991, Balding 1992), including clams (Schell and Kerekes1989)		X
Mussels, snails, leeches (pH >5.0, Schell and Kerekes 1989)		X
The amphipod *Hyalella azteca* (Havens 1994a); pH must remain above 5.8 (Grapentine and Rosenberg 1992)		X
The amphipod *Gammarus minus* (Griffith et al. 1995)		X
Zooplankton		
Some zooplankters, such as *Daphnia galeata mendotae*, *D. retrocurva, Skistodiaptomus oregonensis* (Havens 1993)	X	
The rotifers *Gastropus stylifer, Keratella taurocephala, Polyarthra renata, Symchaeta* sp. (Fore et al. 1996)	X	
The water flea *Bosmina longirostris* (Havens 1993)		X
The rotifers *Asplanchna priodonta, Collotheca mutabilis, Conochiloides* sp., *Conochilus unicornis, Gastropus hyptopus, Kellicota longispina, Keratella*		X

Taxonomic Group and Study Reference	More Tolerant (Less Sensitive)	Less Tolerant (More Sensitive)
cochlearis, Keratella crassa, Polyarthra dolichoptera, Trichocera cylindrica (Fore et al. 1996)		
Functional Feeding Groups		
Scrapers and collectors (Smith et al. 1990)	X	
Shredders (Tuchman 1993)		X
Deposit feeders (Smith et al. 1990)		X

4.9.5 Impacts of Changing Acidity on Habitat for Amphibians and Reptiles

Excessive acidity damages amphibians directly (Horne and Dunson 1994b). Acidity may also have direct impacts as a result of its capacity to mobilize toxic metals and perhaps by making sodium less available in some soil types (Wyman and Jancola 1991).

No studies were found describing the impact of increased acidity on amphibians and reptiles in Washington. Studies from other states, however, document these impacts. The information below summarizes some of the information for amphibian and reptile species that are found in the state, even if the studies were done elsewhere.

In Ontario, the acid-neutralizing capacity (alkalinity) of 38 wetlands positively influenced the probability of the northern leopard frog (*Rana pipiens*) being present (Glooschenko et al. 1992).

Embryos of the tiger salamander (*Ambystoma tigrinum*) had more than 70% survival at pH 4.5 and above but suffered much greater mortality at lower pH levels (Whiteman et al. 1995).

Concerns have been raised regarding the vulnerability to acidification of Montane wetlands in the West. Acidification makes aluminum and cadmium more mobile and increases their concentration in surface waters. Amphibians (e.g., Jefferson's and spotted salamanders) are known to be sensitive to acidity and elevated concentrations of aluminum found in some acidic ponds (Blancher 1991, Ireland 1991, Horne and Dunson 1995).

Aluminum released into Montane pools as a result of acidification sometimes has harmed embryos, reduced growth rates, and/or caused deformities and premature hatching of native amphibians (Bradford et al. 1991, Corn and Vertucci 1992).

4.9.6 Impacts of Changing Acidity on Habitat for Fish

No information was found on the impacts of acidity on fish in Washington's wetlands. In their review of the literature, Adamus et al. (2001) found that acidity can be directly toxic to fish, inhibit reproductive maturation, inhibit spawning behavior, induce emigration,

and alter food availability. Furthermore, in areas where acid rain may be a problem, the increase in acidity induces aluminum toxicity in fish in many lakes and wetlands (Keller and Crisman 1990). Surveys of literature on effects of acidification on fish in lakes (and therefore potentially in wetlands along lake fringes) are provided by Baker and Christensen (1990) and Minns et al. (1990).

4.9.7 Impacts of Changing Acidity on Habitat for Birds

Acidification of wetlands affects birds primarily because it reduces the availability of calcium, which is important for egg development; potentially increases the availability of toxic metals; and alters the species composition and abundance of aquatic insects, submersed plants, amphibians, and fish that are important foods for waterfowl (see previous discussions in Sections 4.9.3, 4.9.4, 4.9.5, 4.9.6).

Changes in the types of available food, especially those rich in calcium, can diminish egg shell thickness and generally reduce the reproductive success of waterbirds in wetlands (Sparling 1990, 1991, Blancher and McNicol 1991, St. Louis et al. 1990, Albers and Camardese 1993). Overall, calcium deficiency appears to affect birds in acidified wetlands more than metal toxicity (Albers and Camardese 1993). Breeding pairs of 15 waterfowl species were more abundant in Ontario wetlands with over 40 parts per million (ppm) total alkalinity than in less alkaline wetlands (Dennis et al. 1989, Merendino et al. 1992). In British Columbia as well, densities of several breeding duck species were greater in ponds with higher levels of conductivity and calcium (Savard et al. 1994).

4.9.8 Impacts of Changing the Acidity on Habitat for Mammals

No information on the effects of acidification on the overall community structure of wetland mammals was located. It can be hypothesized, however, that where acidification becomes severe, community composition may shift from fish-eating species (e.g., otter, mink) to vegetarian or invertebrate-eating species and opportunists (e.g., muskrat, opossum) (Adamus and Brandt 1990).

4.9.9 Summary of Key Points

- No information was found on the impacts of acidity on the hydrologic functions of wetlands, but it is possible to hypothesize that impacts, if any, are minor.
- The rates at which wetlands remove toxic compounds are impacted by increasing acidity because the rates at which denitrification occurs are reduced.
- Increasing the acidity in wetlands can also change the composition of the plant community.

- Impacts of increasing acidity on the habitat provided by wetlands have been documented for invertebrates, amphibians, fish, and birds. In general, increased acidity reduces the richness of invertebrates in wetlands and impacts amphibians either directly or by changing the chemistry of the water in the wetland, making it less suitable as a habitat. Acidic wetlands also become less suitable habitat for birds because the amounts of calcium rich foods are reduced. Mammals are the only group of vertebrates for which no information exists.

4.10 Impacts of Increasing the Concentrations of Salt

Salt concentration in wetlands can increase as a result of (from Adamus et al. 2001):

- Isolating wetlands from some types of groundwater inflow
- Increasing water lost through evaporation
- Discharging effluents (especially irrigation return water)
- Routing runoff that has relatively high conductivity into wetlands

Increased concentrations of salt (salinization) impact the functions of wetlands as described below.

4.10.1 Impacts of Increasing Concentrations of Salt on Hydrologic Functions

No information was found on how changes in salt content might affect the hydrologic functions of flood storage and flood desynchronization. In the absence of any information to the contrary, however, it is possible to hypothesize that salinization will probably not change how wetlands perform these functions. Changes in the salt content of water are not expected to change how well wetlands store water, how well they slow it down during peak flows, or how well they recharge groundwater.

4.10.2 Impacts of Increasing Concentrations of Salt on Functions that Improve Water Quality

One relevant study found that salinities greater than about 300 grams per liter can inhibit the ability of microbes to detoxify toxic forms of selenium (Steinberg and Oremland 1990). This was the only literature found on how salinization might impact the ability of wetlands to remove pollutants.

As noted below, salinization has some impacts on plants, and thus it may affect nutrient uptake and transformation in a wetland. However, it is not possible to predict or hypothesize how such changes in these species might change other functions that improve water quality.

4.10.3 Impacts of Increasing Concentrations of Salt on Plants

In general, high concentrations of soluble salts are lethal to freshwater plants, and lower concentrations may impair growth (Rendig and Taylor 1989). Woody plants tend to be less tolerant than herbaceous plants because they do not have mechanisms for removing salt, other than accumulating salts in leaves and subsequently dropping them (Adamus et al. 2001).

Many plant species that inhabit inland saline wetlands are, of course, adapted to tolerating salt levels that would kill most other wetland plant species. A survey of inland lakes in western Canada which spanned a salinity gradient identified relative tolerance to salinity and specific salinity tolerance thresholds of many wetland species (Hammer and Heseltine 1988).

Individual plant species have different tolerances and reactions to increasing salinity. It can be expected that the plant community in a wetland will change to one dominated by salt-tolerant plants when additional salts are introduced. For example, wetlands in which salt has been present for some time, such as alkali wetlands, have a completely different plant community than that found in non-alkali wetlands. In eastern Washington a major change in plant communities was found when the conductivity (a measure of the amount of salts present in the water) increased to 2.0 milliSiemens and higher (Hruby et al. 2000).

A study by Hutchinson (1991) describes the tolerance of many wetland plants found in Washington. It can be used to predict how the plant species might change in Washington's wetlands as salt concentrations increase.

It can also be expected that wetlands subject to increases in salinity through agricultural practices or discharges of salt will also be subject to a change in plant populations. One wetland undergoing such a change was observed in the Richland area during the calibration of the Washington State Wetland Rating System for Eastern Washington in the summer of 2002. The conductivity of the wetland was measured at about 6.5 milliSiemens. About one-quarter of the area was still dominated by cattails (*Typha latifolia*), a wetland plant with a relatively low tolerance to salt (Hutchinson 1991), but this species was dying. Dead stalks of this species covered almost half the area of the wetland.

4.10.4 Impacts of Increasing Concentrations of Salt on Habitat for Invertebrates

The review of the literature indicates that high levels of salinity can alter the structure of freshwater invertebrate communities in many ways. Adamus et al. (2001) have identified the following mechanisms by which the invertebrate community can be altered:

- Acute and chronic damage to tissues of invertebrates

- Changes in the species composition and structure of algal communities and aquatic plants upon which some invertebrates depend for food and shelter
- Changes in predation on invertebrates by decimating numbers of other crustaceans, fish, and amphibians
- Changes in the bioavailability of some other substances, such as heavy metals and nutrients

Even at low concentrations, increases in chloride (a correlate of salinity, and often associated with road salt applications) among 27 Minnesota wetlands were significantly correlated with declines in species richness among the wetlands (Gernes and Helgen 1999). In Wyoming wetlands of fairly low salinity (0.8 to 30 milliSiemens per centimeter), the dominant macroinvertebrates were amphipods and epiphytic snails. Other recent species-specific salinity data for wetland invertebrates are presented Parker and Wright (1992), and Lovvorn et al. (1999).

4.10.5 Impacts of Increasing Concentrations of Salt on Habitat for Amphibians and Reptiles

In general, relatively little is known about amphibian tolerance to salinity. Three studies have reported a statistically significant negative correlation between conductivity of the water and amphibian species richness (Azous 1991, Platin 1994, Platin and Richter 1995). However, the implications of these studies for understanding impacts on existing populations of amphibians in a wetland that is undergoing an increase in salt concentrations is not clear.

4.10.6 Impacts of Increasing Concentrations of Salt on Habitat for Fish

No information was found on the tolerance of native fishes in Washington to salinity. Adamus et al. (2001) reported the following information relative to some of the introduced game fish that now are found in Washington's wetlands.

Laboratory trials consisting of 120-day exposure of freshwater largemouth bass (*Micropterus salmoides*) to four salinity levels (0, 4, 8, and 12 ppm) indicated a significant decrease in growth rate with increasing salinity up to 8 ppm.

In another experiment, juvenile bluegill (*Lepomis macrochirus*) from a freshwater pond in northeastern Mississippi and a brackish bayou in coastal Mississippi were held in a chamber with zero salinity but given access to chambers containing 0, 2, 4, 6, 8, and 10 ppm salinity (Peterson et al. 1993). Fish from neither habitat showed a clear preference for any of the salinity options. These data and data from previous studies suggest bluegills are better able to physiologically and behaviorally tolerate elevated

salinity relative to other centrarchids (the family of fish containing bluegills, bass, crappies, etc.), particularly bass (Peterson et al. 1993).

4.10.7 Impacts of Increasing Concentrations of Salt on Habitat for Birds

The impacts of increasing salinity on birds are highly dependent on the species in question. The following summarizes relevant studies:

- Highly saline or alkali conditions are detrimental to some invertebrate and plant foods used by many duck species. High salinity is directly toxic or impairs the growth of young ducklings (Clark and Nudds 1991, Moorman et al. 1991).
- Sensitive waterbirds, such as some ducks, may visit saline wetlands but often only when fresher wetlands are available nearby (Lokemoen and Woodward 1992, Woodin 1994, Adair et al. 1996).
- Breeding densities of most duck and grebe species in interior British Columbia were greater in ponds with higher conductivity, but marsh nesting species were unaffected (Savard et al. 1994).

Nonetheless, a few species of water-birds occur regularly at very high densities in alkali wetlands during the breeding season and/or migration. Examples include the American avocet (*Recurvirostra americana*), snowy plover (*Charadrius alexandrinus*), phalaropes, killdeer (*Charadrius vociferus*), horned grebe (*Podiceps auritus*), tundra swan (*Cygnus columbianus*), and white-rumped, semipalmated, and Baird's sandpipers (*Calidris* spp.) (Earnst 1994, Jehl 1994, Savard et al. 1994, Oring and Reed 1997, Rubega and Robinson 1997, Warnock 1997). These relatively salt-tolerant species also occur in less saline wetlands, but their abundance often is greatest in wetlands with high salinity, and is related to sharp seasonal peaks in the abundance of brine shrimp and other salt-tolerant invertebrates. These birds characteristically travel hundreds of miles, sometimes daily or weekly, in order to exploit such invertebrate foods during the short times when the food peaks (Haig et al. 1997).

4.10.8 Impacts of Increasing Concentrations of Salt on Habitat for Mammals

No information was located on the impacts of salinization on the overall structure of mammal communities in wetlands and the suitability of wetlands as habitat for mammals.

4.10.9 Summary of Key Points

- No information was found on the impacts of salinization on the hydrologic functions of wetlands, but it is possible to hypothesize that impacts, if any, are minor.
- Only one study was found that documents any impacts of salinization on the ability of wetlands to improve water quality. Very high salt concentrations inhibit the microbes that detoxify selenium.
- Increasing the salt concentrations in wetlands can change the composition of the plant community.
- Impacts of increased salt concentrations on the habitat provided by wetlands have been documented for invertebrates, fish, and birds. In general, increased salinity changes the composition of the invertebrate community in wetlands. Largemouth bass seem to be especially sensitive to increased salinity relative to other species. The ducklings of some waterbird species may also be sensitive. No information exists on the impact of salinization on mammals and amphibians.

4.11 Impacts of Decreasing the Connection Between Habitats

Decreasing connections between habitats (fragmentation) results directly from human conversion of land to uses that are not part of an undisturbed ecosystem. Fragmentation is a result of both the direct loss of wetlands that isolates populations of wildlife and the creation of barriers to the movement of organisms. Wetland loss and isolation is seen as a major factor contributing to the loss of biological diversity in vertebrate populations that use wetlands (Harris 1988, Gibbs 2000). In general, fragmentation of habitats affects biological diversity through (Harris 1988):

- Loss of the species less tolerant to disturbance or those that inhabit the interior parts of wetlands
- Loss of large species with broad ranges
- Loss of genetic integrity within populations
- Increase in numbers of habitat generalists that thrive in disturbed environments, such as parasites

Occasional migration between wetlands is vital in sustaining local populations of wetland-dependent organisms. Limiting the movements of these species reduces the exchange of genetic material among local populations and can result in population extinctions (Gibbs 2000). Three factors that impede movement among wetlands and other habitats include (Gibbs 2000):

- Greater distances between wetlands
- Degradation of upland habitats
- Increased road density

The effects of fragmentation on wildlife that use wetlands are most extensively documented for amphibians and birds. Little information is available for effects on macroinvertebrates, reptiles, and mammals. Several studies done in the Pacific Northwest are cited in the following discussion of how decreasing habitat connections affects wetland functions.

4.11.1 Impacts of Decreasing the Connection Between Habitats on Hydrologic Functions

Information on how fragmentation affects the flood storage, flood desynchronization, and groundwater recharge performed by individual wetlands was not located in the literature. In the absence of any information to the contrary, however, it is possible to hypothesize that fragmentation will probably not change how individual wetlands still remaining in the landscape perform these functions. Fragmentation at a landscape level is not expected to change how well the remaining individual wetlands store water or how well they slow it down during peak flows. On the other hand, fragmentation probably does impact the delivery and routing of water to wetlands as described in Chapter 3. This may change how much water gets to a wetland for storage but not how well the wetland can store it.

4.11.2 Impacts of Decreasing the Connection Between Habitats on Functions that Improve Water Quality

Information on how fragmentation affects the ability of wetlands to improve water quality was not located in the literature. It is not possible to predict or hypothesize precisely how such changes might affect these functions.

4.11.3 Impacts of Decreasing the Connection Between Habitats on Plants

No information on the response of plant communities to fragmentation was found.

4.11.4 Impacts of Decreasing the Connection Between Habitats on Invertebrates

Few studies were found that documented the impact of decreasing connections on the suitability of wetlands as habitat for invertebrates. One study found that wetland

isolation combined with the harshness of the surrounding upland landscape in more arid environments (such as much of eastern Washington) limit dispersal and colonization by aquatic invertebrates (Myers and Resh 1999).

Another study in New York comparing macroinvertebrate populations at restored wetlands and reference wetlands showed that less mobile invertebrates colonized new wetland sites very slowly or not at all, whereas insects that disperse aerially colonized the new sites rapidly (Brown et al. 1997). Therefore, wetland isolation may have greater effects on less mobile invertebrate species.

4.11.5 Impacts of Decreasing the Connection Between Habitats on Amphibians and Reptiles

4.11.5.1 Amphibians

As early as the mid-1960s, researchers in various parts of the country perceived the effects of reduced connection of habitats on amphibians. One author notes the disappearance of a number of species of frogs, toads, turtles, and snakes in an urbanizing area in the Midwest that he studied from 1949 to 1964 (Minton 1968).

The effects of increased wetland isolation have been extensively studied for amphibians since then. This is probably because amphibians:

- Are restricted to movement on the ground
- Do not typically have large migration ranges
- Often move between terrestrial and aquatic habitats
- Have experienced significant population declines throughout the world

The causes of declines in the populations of amphibians have been extensively studied and most researchers conclude that the problem is very complex and multiple factors are likely at work (Hayes and Jennings 1986, Pechmann et al. 1991, Pechmann and Wilbur 1994, Delis et al. 1996, Adams 1999). Among these factors, there is evidence that increasing isolation of wetlands due to wetland loss may play a significant role in declining amphibian populations (Ostergaard 2000, Adams 1999, Lehtinen et al. 1999, Semlitsch and Bodie 1998). This has significant implications for amphibians in Washington State because about 57% of amphibian species that occur here commonly use wetlands for at least one life cycle stage (Leonard et al. 1993).

Amphibians are not randomly distributed within acceptable habitats—they occur in higher abundance and species richness in habitats that are better connected to other desirable habitats (Lehtinen et al. 1999, Lehtinen and Galtowitsch 2001). A Minnesota study of 21 marshes noted that the two most important predictors of decreases in amphibian species richness in agricultural areas are the degree of wetland isolation and the road density (Lehtinen et al. 1999). The marshes in this study were located in both

prairie and hardwood forest ecoregions in two primary land use settings: urban and agricultural. The study noted some differences between ecoregions and land use effects. In the agricultural prairie ecoregion, the amphibian assemblages observed appeared to be most influenced by:

- Road density
- Wetland isolation
- Biological interactions (presence of predators)

In deciduous forest areas that are urbanizing, amphibian richness was most closely related to upland land use and associated habitat fragmentation.

Other landscape-based studies also conclude that the distances between wetlands, as well as the suitability of terrestrial habitats, are key factors in amphibian distribution. Amphibian recolonization patterns are species and spatially dependent because not all species have the capacity to move beyond fragmented, isolated habitats (Lehtinen and Galatowitsch 2001).

Declines in the richness of amphibian species have also been documented as urban land use increases (Lehtinen et al. 1999, Knutson et al. 1999, Richter and Azous 2001a). A landscape analysis of habitats for anurans (frogs and toads) in Wisconsin and Iowa showed that anurans were positively associated with uplands, wetland forests, and emergent wetlands and negatively associated with urban land (Knutson et al. 1999). A positive association, in this case, means higher abundance and species richness. The negative association with urban land is attributed by the authors to:

- Conversion of habitat
- Roads acting as barriers
- Presence of exotic predators
- Chemical contamination
- Other factors

A study of frog distribution in the Netherlands found that the likelihood of a pond being used by frogs depended on the density of ponds and the amount of suitable terrestrial habitat in the surrounding area (Vos and Stumpel 1995). A similar study in the Netherlands showed that frog use of ponds was negatively correlated with the degree of wetland isolation and road density in the surrounding landscape (Vos and Chardon 1998). Distances between breeding ponds and other life stage habitats, as well as the condition of the terrestrial habitats, were primary factors in determining frog distribution. Open fields were avoided by adults and newly metamorphosed juveniles. Roads increased the mortality of frogs and acted as barriers between wetlands, thus effectively increasing wetland isolation (Vos and Chardon 1998).

Similarly, an Indiana study concluded that amphibian distribution was influenced by (Kolozsvary and Swihart 1999):

- Forest area and proximity
- Density of ponds
- Degree of wetland permanency
- Density of vegetation

The importance of each factor varied for each species.

Using a simulation model, one author concluded that the amount of breeding habitat had a significantly greater effect on the likelihood of population extinction than the extent of habitat fragmentation (Fahrig 1997). Her model showed that if breeding habitat covers more than 20% of the landscape, population extinction is very unlikely no matter how fragmented the habitat. However, this work was based on a generalized model that made a number of assumptions that cannot be verified without targeting a selected species, as do the more empirical studies of amphibian distribution.

Other studies indicate that there is a threshold for extent of wetland isolation or distance between wetlands for each amphibian species. Several studies of maximum distances of amphibian movement to breeding habitats indicate that amphibian reproductive success is affected by wetland isolation and terrestrial habitat condition:

- Richter and Azous (1995) suggest that upland forest habitat must lie within 3,280 feet (1,000 m) of breeding wetland habitat for it to be useful to lentic (pond) breeding amphibian species.
- Baker and Halliday (1999) found limits on the distance that species of newts, frogs, and toads would move to colonize new ponds in England (1,312 feet [400 m] for newts, 3,117 feet [950 m] for frogs and toads). In contrast to other studies, the condition and nature of the adjacent upland habitats did not have a strong correlation to pond colonization. The study may not have been sensitive enough, or the mixed land uses within the agricultural settings may have actually supported amphibian populations.
- The ability of juveniles to move from one wetland to the next depends on the spacing between wetlands and the habitat conditions within the buffers. Distances between ponds directly affect the probability of recolonization and the chance to prevent extinction of amphibian populations. Most individual amphibians cannot migrate long distances and adults return to their home ponds, usually after migrating no more than 656 to 984 feet (200 to 300 m) (Semlitsch 2000).
- A similar study in the Netherlands showed that amphibians would colonize new ponds up to 3,280 feet (1,000 m) away (Laan and Verboom 1990). The authors concluded, however, that the probability of a species colonizing a wetland

increases with proximity to the source wetland and increased connectivity by upland forest habitats between the wetlands.

4.11.5.2 Reptiles

No studies were found that specifically addressed the effects of reduced habitat connections and wetland loss on reptiles. In one study in North Carolina, researchers evaluated the adequacy of federal and state wetland regulations in protecting the habitats that freshwater turtles need to complete their life cycles (Burke and Gibbons 1995). They determined that the area protected as wetland under federal guidelines did not include the area in which two critical life-cycle stages occurred: nesting and terrestrial hibernation. This means that some of the habitats needed for turtle success are vulnerable to loss due to conversion to other land uses. However, this study focused not on the effects of wetland loss but the effects of eliminating upland habitats adjacent to wetlands.

A study that modeled the effects of wetland loss in Maine showed that local populations of freshwater turtles faced a significant risk of extinction following the loss of small wetlands (Gibbs 1993).

As with amphibians, the limited dispersal distances of reptiles, in comparison to birds and mammals, would logically make reptiles particularly vulnerable to habitat fragmentation. However, documentation of the effects of habitat fragmentation on reptiles that use wetlands is very sparse. It appears to be completely lacking for Washington State. This may be due in part to the fact that, with the exception of the western pond turtle (*Clemmys marmorata*), a listed species in the state, no reptile species in Washington are primarily dependent on aquatic habitats. However, the western terrestrial and common garter snakes (*Thamnophis* spp.) are both common near water bodies, including wetlands.

4.11.6 Impacts of Decreasing the Connection Between Habitats on Fish

No information was found on the impacts of fragmentation on the suitability of wetlands as habitat for fish.

4.11.7 Impacts of Decreasing the Connection Between Habitats on Birds

The impacts of decreasing connections between habitats have generally been studied in two types of fragmented landscapes: one fragmented by growing urbanization and one fragmented by agricultural practices. In general there are no studies or conclusions in the literature that would suggest the fragmentation from these two types of land use has significantly different impacts on populations of birds, and so both types of studies are reported below.

The extent of wetland isolation is known to be an important factor that influences bird use of wetland habitats:

- In a study of Puget Sound wetlands, researchers documented a positive association between bird species richness and the proximity of lakes and open water habitats, as well as the structural complexity of the vegetation in the wetlands (Richter and Azous 2001b).
- In northern prairie marshes, bird species richness declined with increased isolation of the wetland (Brown and Dinsmore 1986). Marshes that were part of wetland complexes showed higher species richness than isolated wetlands. Smaller marshes had occurrences of certain bird species only when the marshes were part of a wetland complex.
- These findings are supported by a more recent study of wetland complexes in prairie marshes in Iowa (Fairbairn and Dinsmore 2001). This study related bird species richness and densities of individual species to habitat variables within the wetland complexes and to area of wetland habitat in the surrounding landscape. For some bird species, presence and abundance in a wetland complex were clearly related to the amount of wetland habitat in a 1.9 mile (3 km) area surrounding the complex. A similar study also determined that unfragmented landscapes with prairie marsh supported more waterfowl species than isolated wetlands (Naugle et al. 2001).

The pattern of wetland habitat use varies between different wetland-dependent bird species (Naugle et al. 1999):

- Some species are sedentary and rarely use resources beyond the nest vicinity
- Some use only larger wetlands regardless of the surrounding landscape
- Others require a mosaic of wetlands on the landscape

Therefore, the entire landscape must be assessed, rather than just the wetland patches, in order to determine the habitat suitability of an area for wide-ranging species.

A correlation has been found between the degree of urban development in an area (and the resultant fragmentation) and the extent of declines in native bird species richness. One study in Santa Clara County, California, looked at six sites representing a gradient of development ranging from biological preserve to business district (Blair 1996). Increasing proportions of invasive and exotic bird species were found in the more highly developed areas. The moderately developed sites were highest in species richness and bird biomass. They were, however, lower in native bird diversity than the lesser disturbed sites. The shift in species was related to changes in total available habitat and in habitat structure across the gradient. This study concluded that even relatively minor habitat alterations resulted in loss of species.

Wetlands in the Puget Sound area showed a similar response to urbanization. Researchers found no correlation between total bird species richness and amount of

impervious surface, but there was a correlation with native species richness (Richter and Azous 2001b). The rarer, more sensitive birds, all of which are native, tended to decrease with urbanization. The more adaptive species, with a higher percentage of non-natives (e.g., European starlings [*Sturnus vulgaris*]), tended to increase in urbanizing watersheds. Again, these changes are most likely due to loss of habitat, and therefore reduced connections between habitats, as well as habitat degradation.

One study has important implications because it indicates that duck breeding and brood raising are most successful with a variety of wetlands in close proximity. Conducted in eastern Canada it examined the role that habitat heterogeneity plays in the use of wetlands by ducks (Patterson 1976). It concluded that breeding duck pairs spaced themselves based on the physical size of the wetland. The authors also observed that breeding can occur in relatively "sterile" wetlands (those with hard water). However, duck broods hatched in more sterile wetlands often moved to more biologically productive wetlands where there was a greater food source and more refuge/escape habitat. These preferable wetlands were close to the breeding wetlands because young waterfowl cannot fly.

As with amphibians, the presence of terrestrial habitats between wetlands can be an important factor in waterfowl distribution. A study conducted in an area of intensive wheat farming demonstrates the importance of maintaining connections among habitats for birds (Saunders and DeRebeira 1991). These researchers found that native bird species used corridors as narrow as 13 feet (4 m) to move between patches of preferred habitat. Corridor width was positively correlated with species richness.

A study of bird populations in forest interiors found that habitat fragmentation impairs reproduction and can result in population declines and extinctions (Temple and Cary 1988). Though not focused on wetlands, the study can reasonably be applied to forested wetlands. The authors modeled the effects of habitat fragmentation. They predicted that success rates for nests for forest-interior birds would drop from 70% when nests are greater than 656 feet (200 m) from the forest edge, to only 18% when nests are less than 328 feet (100 m) from the edge. This indicates that fragmentation of forested wetlands through such activities as logging could have significant effects on species that are not tolerant of edge habitats.

In Minnesota, Mensing et al. (1998) assessed the implications of fragmentation at various landscape scales for birds. They found that:

- Diversity and richness of bird species increased with an increase in the extent of forest and wetland within the surrounding landscape.
- Habitats that were in good condition in the areas surrounding wetlands strongly influenced the biotic diversity, with positive correlations shown for birds within 1,640 feet (500 m) of the wetland edge.

4.11.8 Impacts of Increasing Connections Between Habitats on Mammals

Information on the effects of wetland habitat loss and fragmentation on mammals is sparse, even though a number of the mammal species in Washington State are known to commonly use wetlands (beaver, muskrat, mink, otter, water vole, deer mouse, and others). Most of the literature addresses the effects of beaver dams on wetland systems.

One study from the Pacific Northwest documented that wetland fragmentation and the elimination of surrounding upland habitats can have significant effects on small mammals. Richter and Azous (2001c) found that the total area of undeveloped land adjacent to a wetland (including forest, shrub, agricultural fields, and meadows) was weakly associated with mammal richness. A stronger correlation was between the percent of adjacent forest land (within 1,640 feet [500 m] of a wetland) and mammal richness. The highest small-mammal richness was observed in wetlands with at least 60% of the first 1,640 feet (500 m) surrounding the wetland in forest. The authors noted that mammal species richness in Puget Sound wetlands has no correlation with area of impervious surface in the watershed.

Roads are an important factor in habitat fragmentation. For example, a major highway in Massachusetts increased wetland isolation and blocked major travel corridors between suitable habitat patches for mammals (Forman 1998). See Section 4.12.2 for additional discussion of effects of roads on wildlife.

4.11.9 Summary of Key Points

- No information was found on the impacts of fragmentation on the hydrologic functions or the functions that improve water quality.
- Increased wetland isolation appears to be a major factor in species richness and abundance for all taxonomic groups. One author states that "modifications to the environment that preclude movement between component subsystems may be as devastating to vertebrates in the long run as are forces that actually destroy the wetland" (Harris 1988).
- Impacts of fragmentation on the habitat provided by wetlands have been documented for invertebrates, amphibians, reptiles, birds, and mammals. No information was found on impacts to habitat for fish and on the distribution of plants in wetlands.
- Wetland complexes are important to amphibian success. The loss of connections between wetlands has caused reductions in both amphibian abundance and richness.

- The impacts of habitat fragmentation are not as well documented for birds and mammals as they are for amphibians. Certainly there are different issues and patterns of habitat use between these taxonomic groups.

4.12 Impacts of Other Human Disturbances

Human activities on the land create many different types of disturbances. The previous discussion addressed only the major ones that have been studied. The following sections review some of the impacts of other types of disturbances that have been documented to a lesser extent. The discussions in these sections are not separated by wetland function because all of the impacts address either plants or wildlife, and the information is not extensive enough to warrant subdividing it.

4.12.1 Impacts of Altering Soils on Plant Communities

Physically disturbing wetland soils during the dry season, through tillage, compaction, excavation, or other means, can allow invasion by non-native plant species (Morin et al. 1989, Sutton 1996, David 1999, Galatowitsch et al. 1999). It can also destroy much of the viable seed bank (Lee 1991). Tilling the soil often reduces diversity, including both richness and evenness, as documented in a Carolina bay wetland (Kirkman and Sharitz 1994). The tillage treatment disrupted the roots of perennials more than burning, and it encouraged germination of annuals in the seed bank and colonization by several invasive species.

Invasive plants, especially non-native plants, significantly alter the species composition of many wetlands, sometimes even forming nearly monotypic stands. Among the most widespread invaders in North America are cattail (*Typha*), reed canarygrass (*Phalaris* sp.), purple loosestrife (*Lythrum salicaria*), giant reed (*Phragmites* sp.), milfoil (*Myriophyllum spicatum*), and hydrilla (*Hydrilla verticillata*). Their increased dominance is frequently attributed in part to the physical disturbance of soils or water levels within a wetland and/or the surrounding landscape, including accelerated sedimentation, eutrophication, and the construction of mitigation wetlands (Confer and Niering 1992, Magee et al. 1999).

Continuously disturbing the soil, for example through compaction and road building, can alter species composition. These disturbed conditions can lead to a decline in both the biomass of native species and a change in the soil conditions that support them (Ehrenfeld and Schneider 1991). Use of all-terrain vehicles also impacted wetlands on the Atlantic coastal plain, reducing the density of seed in wetland seed banks and allowing common rushes to displace rare species (Wisheu and Keddy 1991). Excavation and clearing of gas pipeline rights-of-way through forested wetlands in Florida resulted in increased species richness within the wetland clearings but an increased percent cover of non-native species (van Dyke et al. 1993).

4.12.2 Impacts of Roads on Wildlife

Roads contribute to lower species richness for a variety of wildlife groups through the factors listed below. While all of the studies cited in this section were conducted in other regions of the country, much of the information is likely to pertain to effects on Pacific Northwest wildlife because the effects are inherent to roads regardless of region.

It is theorized that roads cause the loss of biodiversity by (Findlay and Bourdages 2000):

- Restricting movement between populations of wildlife
- Increasing mortality
- Fragmenting habitat
- Increasing edge habitat
- Facilitating invasion by exotic species
- Increasing human access to wildlife habitats

Findlay and Bourdages (2000) note that there may be long time lags between road construction and the time when effects on wildlife are perceptible. Effects may be undetectable in some taxa for decades.

Increased road density is implicated in lower species richness and abundance for vertebrates. In wetlands in southeast Ontario, the species richness of mammals, amphibians, reptiles, and birds was seen to decline with increased road density (Findlay and Houlahan 1997). Road construction and forest removal are viewed by these authors as increasing the risk of loss of biodiversity in wetlands.

Frog and toad density decreased with increasing traffic in another study by Fahrig et al. (1995). This study concluded that increased road density can contribute to amphibian population declines in urbanizing areas. A study of amphibians using small isolated wetlands in Florida found high mortality during migration between upland terrestrial habitats and temporary pond breeding habitats (Means 1996). The author attributes much of this to direct road mortality.

A study of the “road-effect zone” of a four-lane suburban highway in Massachusetts was undertaken to determine the distance from a road that impacts can be measured (Forman 1998). This study concluded that the road blocks migration routes for salamanders up to several hundred meters from wetlands. The study also showed that the effect of the road on blocking major travel corridors between suitable habitat patches for small mammals could be measured to several kilometers from the road. The effects of traffic noise on birds could be measured up to 2,132 feet (650 m) from the road in forested areas and 3,051 feet (930 m) in open areas. The implications of effects on wetland wildlife are evident even though the findings of this study are applicable to a variety of habitats. (See Section 4.12.3 for more on the effects of noise on wildlife.)

A related study of the same Massachusetts highway showed that significant ecological effects extended out at least 328 feet (100 m) from the highway. Forman and Deblinger (2000) studied nine ecological factors relating to, among other things, wetlands, streams, and amphibians. Assessing all factors, this study concluded that the "road-effect zone" averaged approximately 1,969 feet (600 m) wide, though it was quite variable in width at specific locations.

4.12.3 Impacts of Noise on Wildlife

The effect of noise on wildlife is a topic of growing concern. The frequency of the sound waves and the duration of the sounds influence how noise affects wildlife species. Although many of the studies discussed below do not address wetlands specifically, the impacts of noise are not expected to change whether the species in question is in a wetland or another type of habitat.

Frequency is the perceived pitch of sound, and different animals show different sensitivities to the same range of frequencies. Generally, smaller mammals such as rodents, shrews, and bats have a greater sensitivity to higher frequencies—often within ranges exceeding 20,000 Hertz (Hz), the upper limit of human sound perception. Larger mammals show sensitivity to low frequencies and may be able to detect sound at or below 10 Hz. While most birds show auditory sensitivity similar to humans (20 to 20,000 Hz), certain birds (such as rock doves) can also perceive low-frequency sounds, often with much greater sensitivity than their larger mammalian counterparts (Kreithen and Quine 1979). Some frogs and toads also show low-frequency sensitivity (Hetherington 1992), and even some small mammals are capable of discerning sounds of only a few Hertz (Plassman and Kadel 1991).

Sound duration may be divided into two classifications: continuous sounds which last for a long time with little or no interruption, and impulse sounds lasting for only short durations (Larkin et al. 1996). Impulse sound and continuous sound appear to have different physiological and behavioral effects. Generally, impulse noise appears to be more stressful to wildlife, at least in part due to the unpredictability of such noise (Larkin et al. 1996).

Overall, the literature suggests that species differ widely in their physiological response to various types, durations, and sources of noise (Manci et al. 1988). However, noise effects on wildlife may be broadly classified as primary, secondary, and tertiary:

- **Primary** effects are direct, physiological changes to the auditory system and may be considered to include the "masking" of auditory signals. Masking is the inability of an individual to hear important environmental signals such as calls from mates or noises of predators or prey.
- **Secondary** effects may include non-auditory physiological effects such as stress and hypertension, as well as behavioral modifications that include interference with mating or reproduction and impaired ability to obtain adequate food, cover, or water.

- **Tertiary** effects are the direct result of primary and secondary effects at a population level and include population decline and habitat degradation. Most of the effects of noise are mild enough that they may never be detectable as variables of change in population size or population growth against the background of normal variation (Bowles 1995).

The behavioral responses of wildlife to noise show a high degree of variation depending on the species, the type of noise, and the habituation of the individuals to the source of noise. For example, some bald eagles can be very tolerant of auditory stimuli when the sources are screened from view (Stalmaster 1987), but other raptor species such as prairie falcons flush from perches and nests at sudden loud noises (Harmata et al. 1978).

Animals may become tolerant of repeated noises. Krausman et al. (1986) studied desert ungulates exposed to aircraft noise and noted that short-term habituation to aircraft noise occurred with repeated exposure. Sandhill cranes nesting meters away from a Florida highway showed no response to passing traffic (Dwyer and Tanner 1992). The effects of noise vary not only with the type of noise in question, but with an individual animal's experience, time of day (Herbold et al. 1992, Gese et al. 1989), and reproductive cycle (Platt 1977).

Research on the effects of traffic noise on breeding birds was conducted by Reijnen et al. (1995, 1996) who studied woodland and grassland bird populations in the vicinity of roadways. Ambient noise up to a given level resulted in no reduction in the density of bird populations. However, once an ambient noise threshold level was exceeded, densities decreased exponentially with increased noise. Threshold levels were found to range from 36 to 58 decibels, depending upon species, and the zones of decreased breeding densities surrounding the roadways ranged up to 2,670 feet (810 m) for particularly sensitive species near busy roadways. They found habitat avoidance by individual birds in habitat that would otherwise have been suitable for breeding.

Research on amphibians also found evidence that reproductive output may be diminished in frogs breeding near highways because of acoustic interference (Barass 1985 in Larkin et al. 1996).

4.12.4 Impacts of Recreational Activities on Wildlife and Plant Communities

The effects of recreational activities in wetlands do not appear to be very well studied given the lack of recent articles on this topic. Most of the available information is anecdotal and focused on the more evident impacts such as loss of vegetation from the use of off-road vehicles. There is less information on the effects on wildlife of such disturbances as noise, light, glare, and human presence caused by recreational activities, particularly with respect to wetlands. None of the studies described in this section were located in the Pacific Northwest.

A synthesis paper on management of amphibians in Montana notes that among the many factors that are likely to contribute to a decline in amphibian populations are trail development, on- and off-road vehicle use, and development and management of recreational facilities (Maxell 2000). Citing a number of studies from the 1980s, Klein (1993) notes that recreational uses in natural areas can disrupt:

- Wildlife foraging and social behavior
- Animals that are feeding
- Parent-offspring bonds
- Pair bonds

The author also cites several studies stating that increased predation of nests and decreased densities of wildlife result from greater human recreational use of natural areas.

A study of flooded gravel pits in Britain examined the abundance and distribution of one species of wintering waterfowl with regard to recreational disturbance (Fox et al. 1994). The authors found that water-based recreational activity, such as boating, reduced the number of birds on the ponds to the greatest extent of all the observed activities. Ponds where fishing, walking, or other bank-side activities were allowed also showed reduced numbers of birds in comparison to the ponds that were designated reserves with restricted access. They were, however, not as reduced in abundance as those ponds that also allowed water-based activities.

The effects of recreational use on waterfowl were also studied in a nearshore area on Lake Erie (Knapton et al. 2000). Excessive human disturbance reduced the foraging efficiency and body fat acquisition for waterfowl and can result in decreased bird densities. Diving ducks appeared to be the most sensitive to disturbance. Furthermore, recreational shooting poses additional threats to wildlife if lead shot is used. The impacts of lead shot are discussed in Section 4.8.8.

In another study on recreation impacts on birds, Klein (1993) studied the specific behaviors of humans that disturb wildlife on a subtropical barrier island that is a National Wildlife Refuge off the coast of Florida. Her study sites were primarily in mudflat and mangrove wetland habitats. She tested a variety of treatments such as driving by without stopping, stopping the vehicle with and without getting out, approaching the birds on foot, and playing noise tapes. The author found that most of the bird species present were disturbed by the noise tape. Some species such as great blue heron consistently flew away when approached by a person, whereas other species tolerated human presence until closely approached.

Klein (1993) concludes that car traffic is less disruptive to wildlife than out-of-vehicle activity. Frequent human approaches may cause some bird species to forage in areas with fewer intrusions. Wildlife photographers were the most likely visitors to approach birds. Visitors who spoke with refuge staff and volunteers were the least likely to disturb

birds, possibly due to an increased awareness of the needs of wildlife. While this study involved a very different ecosystem, it is useful because it generated data on bird species that also occur in Washington. It also is one of the few studies that examined the effects of specific human behaviors on wildlife.

Recreational activity is believed to be one of the main factors in lakeshore deterioration and decline in reed-dominated wetlands in a study of Central European lakes (Ostendorp et al. 1995). It is likely that trampling of bank-side vegetation by recreationalists is causing bank erosion and excessive siltation in nearshore wetlands.

Although recreation often occurs in more rural habitats, urbanization also brings increased intensity of recreational uses within remaining greenbelts and open spaces. A study in western Australia examined the trend in smaller lot size relative to the owners' use of nearby open spaces (Syme et al. 2001). Smaller lot size resulted in an increase in recreational visits by the homeowners to nearby wetlands. Increased access to and recreational use of wetlands is clearly one of the impacts that accompany urban development.

4.12.5 Impacts of Invasion by Exotic Species

Urban, suburban, and agricultural developments increase the likelihood of introducing exotic animal and plant species to wetlands. The following factors have been found to increase the opportunity for introducing exotic species:

- Increased access to wetlands through higher road densities
- Greater fragmentation of the landscape
- Higher densities of human land use
- Alterations of wetland hydroperiods
- Direct disturbance of wetlands

Exotic wildlife that has been introduced to the Pacific Northwest affects wetlands and wetland wildlife. The studies cited in the following discussion implicate disease, predation, and competition as major factors in limiting the success of native wildlife. While some causal relationships are clear, such as starlings displacing cavity-nesting ducks, others are less understood.

4.12.5.1 Impacts of Exotic and Invasive Plants in Wetlands

Invasive plants, especially non-native invaders, significantly alter the composition of plant communities in many wetlands, sometimes even forming nearly monotypic stands (Adamus et al. 2001). Changes in the plant community can be expected to result in changes to all the invertebrates and microscopic organisms that are associated with specific plant species.

Among the most geographically widespread invaders in Washington's wetlands are reed canarygrass (*Phalaris arundinaceae*), purple loosestrife (*Lythrum salicaria*), giant reed (*Phragmites* sp.), and European milfoil (*Myriophyllum spicatum*). Their increased dominance is frequently considered to be a result of human disturbances such as the following:

- Changes in soils or water levels within a wetland and/or the surrounding landscape, including accelerated sedimentation, eutrophication, and the construction of mitigation wetlands (Confer and Niering 1992, Magee et al. 1999)
- Changes in hydroperiod following urbanization (Cooke and Azous 2001)
- Increased human access and mechanical disturbance of wetlands (e.g., a study in southern Australia showed that vegetation removal and site disturbance are major factors in plant invasions; Detenbeck et al. 1999)

4.12.5.2 Impacts of Domestic Pets

Residential development typically brings increased access to wetlands by domestic pets, primarily cats and dogs. A study of house cat predation in Australia indicates that small mammals were the preferred prey of house cats, but cats also killed birds, reptiles, and amphibians (Barratt 1997). Many of the mice and rats collected by the cats in this study are exotic species themselves, but the results suggest that house cats may have significant impacts on native populations as well, particularly along the fringes of suburban expansion where native mammals are more common.

A similar study of house cat predation in Virginia determined that individual cats caught an average of 26 prey in urban areas and 83 prey in rural areas over an 11-month period (Mitchell and Beck 1992). Extrapolating these numbers of prey individuals to the total number of cats in a specific urban or suburban area would give an astonishingly high prey toll related to house cats. Taxonomic groups that were preyed upon in this study included birds, mammals, and reptiles.

4.12.5.3 Impacts of Exotic Wildlife

In Washington and Oregon about 42 exotic vertebrate species have established populations (Witmer and Lewis 2001). These include species of 18 birds, 19 mammals, three reptiles, and two amphibians. The birds were mainly introduced for hunting or aesthetic purposes, while the mammals mostly escaped from commercial or domestic settings. The amphibians and reptiles were released pets or were introduced for food or aesthetic purposes. About 30% of these species are restricted to freshwater and riparian systems, although others among this group will commonly use these habitats.

Some of the ecological consequences of these introductions for wetlands and wetland wildlife are well documented. Many introduced birds are known to usurp nests of native birds or to compete with them for nest sites. European starlings (*Sturnus vulgaris*) are known to displace wood ducks, woodpeckers, and other species from their nests, often

destroying the eggs and young (Witmer and Lewis 2001). Starlings also out-compete many native species for nest cavities, overwhelming them with their large numbers and aggressive behavior. Transmission of disease, particularly from exotic birds and Old World rodents, is also a major problem that threatens native wildlife (Witmer and Lewis 2001).

Introduced mammals affect native wildlife and plants through predation and herbivory (Witmer and Lewis 2001). For example, nutria (*Myocaster coypus*), which were introduced from South America for fur production, have tremendous impacts on wetland vegetation, uprooting plants as they dig for rhizomes and denuding vast areas (Mitsch and Gosselink 2000, Witmer and Lewis 2001). Nutria may be implicated in population declines of muskrats (*Ondatra zibethicus*), probably due to competitive exclusion (Witmer and Lewis 2001).

4.12.5.4 Impacts of Exotic Invertebrates in Wetlands

Humans have introduced a number of non-native invertebrates to wetlands. Native invertebrate communities seem ill-adapted to compete with or avoid these alien species, but data on long-term effects to wetland communities are mostly lacking.

The zebra mussel (*Dreissena polymorpha*) has invaded many aquatic systems throughout North America (d'Itri 1997). This species can totally carpet substrates, displacing native mussels (Tucker and Atwood 1995), as well as some midges, snails, and caddisflies. The mussel has minimal or positive effects on amphipods and flatworms (Wisenden and Bailey 1995). They may also concentrate contaminants, making them more available to invertebrate food chains (Bruner et al. 1994). The rapid spread of zebra mussels may have been made more possible by the preceding decline of native mussels as a result of pollution and changes in habitat (Roberts 1990, Nalepa and Schloesser 1993, Hebert et al. 1991, Mackie 1991, Haag et al. 1993, Whittier et al. 1995).

Because unionid mussels in rivers are relatively immobile and have long life spans (often over 10 years), they are particularly susceptible to disruptions from introduced mussels as well as from impoundments and channelization (Mehlhop and Vaughn 1994). Riverine wetlands with higher alkalinity tend to be more susceptible to invasions by zebra mussels (Whittier et al. 1995). Wetlands along rivers might serve as refuges for native mussels otherwise impacted by expansion of the zebra mussel population (Tucker and Atwood 1995).

4.12.5.5 Impacts of Exotic Amphibians in Wetlands

The effects of exotic species of amphibians on native amphibians that use wetlands are particularly well studied. Predation and competition from introduced amphibians has been suggested as one cause of population declines for native amphibian species (Witmer and Lewis 2001).

Bullfrogs (*Rana catesbeiana*) are often cited as a factor in declining amphibian populations (Hecnar and M'Closkey 1996, Kiesecker and Blaustein 1998, Adams 1999).

Native to eastern North America, bullfrogs were introduced to the Pacific Northwest in the early 1900s for hunting and food. The species establishes easily in the wild because it can colonize a variety of aquatic habitats and is a prolific breeder. Bullfrogs are suspected of causing amphibian declines because they prey on frogs and salamanders and are often so numerous in wetlands that they are thought to out-compete native species for space (Witmer and Lewis 2001).

Studies of the role that bullfrogs play in declines of amphibian populations are, however, somewhat contradictory in their findings (Hecnar and M'Closkey 1996, Kiesecker and Blaustein 1998, Adams 1999, Witmer and Lewis 2001). It is possible that the effects of bullfrogs may differ for various species, or their influence may be quite subtle and complex.

Furthermore, several studies conducted in the Pacific Northwest have found either weak or no correlation between bullfrog presence and amphibian richness and abundance (Adams 1999, Richter and Ostergaard 1999, Richter and Azous 1995). Data from a monitoring program of amphibians in King County wetlands showed that bullfrogs are not causing competitive exclusion of native species (Richter and Ostergaard 1999). Native amphibian richness was not negatively correlated with bullfrog presence or with the presence of permanent water in the wetlands (Richter and Ostergaard 1999). Richter and Azous (1995) noted relatively high species richness for native amphibians in permanently ponded wetlands, the preferred habitat for bullfrogs.

Focusing on red-legged frogs (*Rana aurora*) in Puget Lowland wetlands, Adams (1999) concluded that this species is not excluded from wetlands that also support bullfrogs. The study showed little to no negative correlation between red-legged frogs and bullfrogs. It noted that exotic fishes such as sunfish, yellow perch (*Perca flavescens*), and smallmouth bass (*Micropterus dolomieui*) had greater effect on amphibian richness in the wetlands studied.

A study of red-legged frogs in the Willamette Valley in Oregon, however, stated their development was affected by both bullfrogs and exotic fishes (Kiesecker and Blaustein 1998). In this study, tadpoles showed decreased mass at metamorphosis and increased time to metamorphosis in the presence of larval and adult bullfrogs. Smallmouth bass alone had little effect on tadpole development, but red-legged frog tadpoles altered their use of microhabitats when both bullfrogs and smallmouth bass were present. Survival of tadpoles was affected only when both bullfrog adults and larvae were present, or when both bullfrog larvae and smallmouth bass were present.

Leonard et al. (1993) surveyed populations of the northern leopard frog (*Rana pipiens*) in Washington State. They found that the species had been extirpated from most of its historic range, with only small populations remaining in parts of eastern Washington. These authors noted that areas once inhabited by the northern leopard frog support exotic species, including bullfrog and such fish species as largemouth bass (*Micropterus salmoides*), black crappie (*Pomoxis nigromaculatus*), yellow perch, and brown bullhead. They theorized that these species may be implicated in the decline of the northern leopard frog but have no definitive data to support this hypothesis.

In a review of studies across the country on potential causes of frog population declines, Hayes and Jennings (1986) concluded that existing studies do not support the theory that bullfrogs are a major cause. They argue that predation by exotic fishes may be a more likely hypothesis but note there is little data to support this. However, studies in the Pacific Northwest appear to support this theory (Adams 1999, Aker 1998). A study in the Okanogan Highlands in northeast Washington showed that richness of pond-breeding amphibian and abundance were diminished by the presence of exotic fish (Aker 1998). The non-native fish species observed in this study included largemouth bass, tench (*Tinca tinca*), brook trout (*Salvelinus fontinalis*), and perch. While there was lower amphibian richness in ponds with native fish than those with no fish, the data indicate that non-native fish had a greater impact on amphibian numbers and richness.

4.12.5.6 Impacts of Exotic Fish in Wetlands

Non-native fish have been widely introduced into waters of the United States, both intentionally and by accident. Adamus et al. (2001) cite research showing that the effects of invading species on native fish communities are usually adverse (Baltz and Moyle 1993), especially when coupled with simultaneous impacts from other factors (Larimore and Bayley 1996, Marschall and Crowder 1996).

The introduction of carp has resulted in significant impacts on wetlands in eastern Washington. Large, herbivorous fish such as carp compete directly with birds for submerged aquatic plants (Bouffard and Hanson 1997). The fish also resuspend the sediments on the bottom of lakes and ponds, and this has a significant impact on invertebrates as well as the submerged aquatic plants (see Section 4.5.5).

4.12.6 Summary of Key Points

- Alteration of soils can change the plant community in a wetland and allow invasion by exotic species.
- Noise creates stress for wildlife, but the impacts are very specific to individual species and to the type of noise generated.
- Recreational use of wetlands impacts the normal behavior of wildlife and reduces densities.
- Invasions by exotic species can alter the distributions of both plant and animal species in wetlands. The impacts of bullfrogs on other amphibians, however, are ambiguous even though this question has been studied extensively.

4.13 Chapter Summary and Conclusions

Humans create many different types of disturbances that can affect the environmental factors that control the performance of wetland functions. These disturbances were

reviewed in Chapter 3. Chapter 4 has reviewed the information available on how these human disturbances impact wetlands and their functions. The disturbances that impact wetlands the most include:

- Direct changes to the physical structure of wetlands via filling, vegetation removal, tilling of soils, and compaction of soils
- Changes in the amount of water in wetlands
- Changes in how water levels fluctuate (frequency, amplitude, direction of flows)
- Changes in the amount of sediment
- Increases in the amount of nutrients
- Increases in the amount of toxic contaminants
- Changes in the amount of acidity
- Increasing the concentration of salts
- Decreasing the connection between habitats
- Other disturbances that are not as well documented including alteration of soils, construction of roads, noise, recreational access, and invasion of exotic species

Table 4-3 reviews how various land use practices create disturbances that can change the environmental factors that control wetland functions. Table 4-4 summarizes the effects of each of these disturbances in terms of the wetland functions they may impact. The rating of the impacts in the table represents a synthesis by the authors of all the information presented in this chapter. By combining the information in these two tables, it is possible to associate changes in functions of wetlands with general types of human land use, as shown in Table 4-5.

For example, Table 4-3 shows that urbanization creates significant disturbances that change the amount of water, the fluctuations of water levels, and input of sediments, nutrients, and contaminants to wetlands. Table 4-4 shows that disturbances to water flows, fluctuations of water levels, and input of sediments, nutrients, and contaminants have a significant impact on the wetland functions of providing habitat for plants, invertebrates and reptiles/amphibians. Table 4-5 synthesizes this information to show that urbanization impacts the habitat for plants, invertebrates, reptiles, and amphibians in wetlands. The human land uses create various disturbances in the environment, and those disturbances in turn affect the factors that control wetland functions, ultimately leading to changes in those functions.

The scientific information available indicates that human activities and uses of the land can have significant impacts on the functions in wetlands at both the larger, landscape scale and at the scale of the individual wetland itself. As a result many different approaches and methods have been developed to try to minimize these impacts. These

methods include regulations to control human activities near wetlands, methods to replace the functions, and ways to protect the wetland resource through restoration. The effectiveness of some these tools at actually protecting wetland functions are discussed in Chapters 5 and 6.

Table 4-3. Disturbances resulting from different land use practices that can change the factors that control wetland functions.

Key to symbols used in table:

(xx) land use creates a significant disturbance of environmental factors

(x) land use creates a disturbance

(nm) studies on impacts of this land use do not mention this disturbance

(h) literature is lacking but disturbances can be hypothesized based on authors' experience

(?) information lacking

Disturbance	Scale of Disturbance	Agriculture	Urbanization	Mining
Changing the physical structure within wetlands (filling, vegetation removal, tilling of soils, compaction of soils)	Site scale	xx	xx	h
Changing the amounts of water	Landscape scale Site scale	xx xx	xx xx	? h
Changing fluctuations of water levels (frequency, amplitude, direction of flows)	Landscape scale Site scale	xx xx	xx xx	? h
Changing the amounts of sediment	Landscape scale Site scale	xx xx	xx xx	h h
Increasing the amount of nutrients	Landscape scale Site scale	xx xx	xx xx	nm nm
Increasing the amount of toxic contaminants	Landscape scale Site scale	xx xx	xx xx	x xx
Changing the acidity	Landscape scale Site scale	nm nm	nm nm	x xx
Increasing the concentrations of salt	Landscape scale Site scale	x x	nm nm	nm nm
Decreasing the connection between habitats	Landscape scale	xx	xx	h
Other disturbances	Site scale	xx	++	h

Table 4-4. Synthesis of the information reported in the literature on the impact of different human disturbances on wetland functions.

Key to symbols used in table:

++ Significant impacts on specific functions have been documented
+ Some data suggest impacts or impacts could be hypothesized
0 Data indicate that impacts are minimal
? Information is lacking

Disturbance Type	Functions							
	Hydrologic	Water Quality	Plants	Habitat for Invertebrates	Habitat for Amphibians and Reptiles	Habitat for Fish	Habitat for Birds	Habitat for Mammals
Changing the to physical structure of wetland	+	+	++	++	+	+	++	+
Changing the amount of water	+	+	++	++	++	+	+	?
Changing fluctuations of water levels	?	?	++	+	++	+	?	?
Changing amounts of sediment	+	?	++	++	?	?	?	?
Increasing amounts of nutrients	+	+	++	++	++	+	+	+
Increasing amounts of toxic contaminants	?	+	++	++	++	++	++	?
Changing acidity	0	+	+	++	++	+	+	+
Increasing concentrations of salt	0	?	++	++	?	?	+	?
Decreasing connections between habitats	0	?	?	?	++	?	++	+
Other disturbances	?	?	++	+	++	++	++	++

Note: A (++) does not indicate the direction of the impacts to functions. In some cases the disturbance can increase the function or the richness and abundance of species and in other cases it can decrease them. A disturbance can also decrease or increase a function depending on the intensity of the disturbance (e.g., small amounts of nutrients can increase invertebrate richness and abundance, but too much will cause eutrophication).

Table 4-5. Synthesis of the impacts of different land uses on wetland functions.

Key to symbols used in table:

++ Significant impacts on specific functions have been documented
+ Some data suggest impacts or impacts could be hypothesized
? Information is lacking
+? Some impacts have been documented but more information is needed

Land Use	Functions							
	Hydrologic	Water Quality Improvement	Plants	Habitat for Invertebrates	Habitat for Reptiles and Amphibians	Habitat for Fish	Habitat for Birds	Habitat for Mammals
Agriculture	+	+	++	++	++	++	++	+?
Urbanization	+	+	++	++	++	++	++	+?
Mining	?	?	+	++	++	+	+	+?

Note: A (++) does not indicate the direction of the impacts to functions. In some cases the land use can increase the function or the richness and abundance of species and in other cases it can decrease them. A land use can also decrease or increase a function depending on the intensity of the land use.

Chapter 5
The Science and Effectiveness of Wetland Management Tools

5.1 Reader's Guide to this Chapter

This chapter builds on the previous discussion of how wetlands function (Chapter 2), how human activities and changes in land use cause disturbances (across the landscape and at specific sites) that influence the factors that control wetland functions (Chapter 3), and how wetland functions are impacted by these disturbances (Chapter 4).

Chapter 5 presents a synthesis of what the current literature reports on four tools currently used to address impacts to wetlands and their functions: wetland definitions, wetland delineation methods, wetland ratings, and buffers. This chapter does not provide language or recommendations for regulatory or policy language—those will be provided in a separate volume on management options and recommendations (Volume 2).

5.1.1 Chapter Contents

Major sections of this chapter and the topics they cover include:

Section 5.2, Introduction and Background on Regulatory Tools introduces the key wetland management tools that are discussed in this chapter.

Section 5.3, How Wetlands Are Defined and Delineated describes similarities and differences in the way various agencies define what a wetland is. It explains the critical difference between *biological wetlands* and *regulated wetlands* and discusses certain types of wetlands that are frequently exempted from regulation such as isolated wetlands, small wetlands, or those designated as Prior Converted wetlands. The various manuals that have been developed to guide the delineation of wetland boundaries are then briefly discussed.

Section 5.4, Wetland Rating Systems discusses how rating systems have been developed to rapidly assess wetland characteristics in the field. These characterizations allow wetlands to be rated for regulatory or management purposes. The section introduces the reader to the Washington State Wetland Rating System, which was briefly mentioned previously in a number of places in the document. It also includes discussion of certain wetland types that are singled out for special attention under the Washington State rating system.

Section 5.5, Buffers comprises the bulk of this chapter. This section provides a synthesis of the literature on how buffers protect and maintain wetland functions. The section concludes by summarizing recommendations from the literature for establishing effective buffer widths.

Section 5.6, Chapter Summary and Conclusions ties together the major concepts presented in the chapter.

5.1.2 Where to Find Summary Information and Conclusions

Each major section of this chapter concludes with a brief summary of the major points resulting from the literature review on that topic in a bulleted list. The reader is encouraged to remember that a review of the entire section preceding the summary is necessary for an in-depth understanding of the topic.

For summaries of the information presented in this chapter, see the following sections:

- Section 5.3.6
- Section 5.4.2
- Section 5.5.3.5
- Section 5.5.4.4
- Section 5.5.5.4
- Section 5.5.6.1

In addition, Section 5.6 provides a summary and conclusions about the overarching themes gleaned from the literature and presented in this chapter.

5.1.3 Data Sources and Data Gaps

No literature review was conducted for the section on wetland definitions or delineations. Both of these management tools are currently established by state and federal statutes, and it was determined that review of the previous discourse on those topics was not relevant to the current state of the science for Washington State. Several synthesis documents on small and isolated wetlands have been published since 2000. Considerable research was published prior to 2000, focused on the role of small wetlands relative to wildlife in a landscape context.

Papers on the adequacy or effectiveness of wetland rating systems were not found; instead the literature focuses on function assessment methods. This chapter does not attempt to assess the science on wetland function assessment because the Washington Department of Ecology has completed function assessment methods for several different

wetland hydrogeomorphic types on both sides of the state within the last five years (see Chapter 2 for further information).

The subject of buffers is well documented in the scientific literature. Literature related to agricultural practices and vegetated filter strips from the north-central United States and south-central Canada is relevant to some agricultural practices in Washington, especially in areas east of the Cascades. Studies on buffers in urban and suburban settings conducted in the Pacific Northwest are also relevant.

The majority of research on buffers tends to focus on how buffers influence water quality. Far fewer studies examine the influence of a buffer's physical characteristics on attenuating rates of surface water flow.

Most studies on buffers related to wildlife document the needs of a particular species or guild related to how far they travel from aquatic habitats to fulfill their life-histories. While there is substantial literature on the implications of habitat fragmentation and connectivity, this literature does not specifically address the role of buffers in providing connectivity between wetlands and other parts of the landscape.

Numerous compilations and syntheses of "buffer" literature have been completed since 1990. These synthesis documents are used in this report as direct sources when no more recent research was found. This chapter also cites literature related to stream buffers and riparian areas when the findings are relevant to the functions or processes these areas provide to the adjacent aquatic resource.

A more detailed description of the types of literature used and any recognized gaps in the scientific literature is provided within each section on buffers as appropriate.

5.2 Introduction and Background on Regulatory Tools

The regulatory tools discussed in this chapter are components of "typical" wetland protection programs. The intent is not to analyze all elements of protection programs and their regulations but to focus on the key science-based elements relating directly to wetland protection and management. Therefore, this chapter focuses on the following four elements:

- Wetland definitions
- Wetland delineation methods
- Wetland ratings
- Buffers

The topic of compensatory mitigation, another key regulatory tool, is discussed separately in Chapter 6 because of the volume of information and literature available on this subject.

5.3 How Wetlands are Defined and Delineated

5.3.1 How Agencies Define Wetlands

Several definitions of wetlands have been developed and used by various federal, state, and local agencies and jurisdictions. The effectiveness of current federal or state wetland definitions was not evaluated as part of this report. However, definitions are included here because how a wetland is defined is critical to determining what areas are subject to the provisions of a law or regulation.

For the purposes of most laws and regulations, wetlands are usually defined using one of the following two definitions:

> *Those areas that are saturated or inundated by surface or groundwater at a frequency and duration sufficient to support, and under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs, and similar areas.* (U.S. Army Corps of Engineers 1987);
> or

> *"Wetlands" or "wetland areas" means areas that are inundated or saturated by surface or ground water at a frequency and duration sufficient to support, and that under normal circumstances do support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas. Wetlands do not include those artificial wetlands intentionally created from non-wetland sites, including, but not limited to, irrigation and drainage ditches, grass-lined swales, canals, detention facilities, wastewater treatment facilities, farm ponds, and landscape amenities, or those wetlands created after July 1, 1990, that were unintentionally created as a result of the construction of a road, street, or highway. Wetlands may include those artificial wetlands intentionally created from nonwetland areas to mitigate the conversion of wetlands.* (Washington Administrative Code 173-22-030.)

The Washington State definition is derived from the U.S. Army Corps of Engineers definition, but it also includes clarifying language that identifies which common human-made or induced features are not meant to be defined as wetland. The state definition is required by the Growth Management Act (RCW 36.70A.030 (20)) to be used in all local critical area regulations.

In addition, for the process of conducting the National Wetland Inventory, the U.S. Fish and Wildlife Service (USFWS) defined wetlands as follows:

> *Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For the purpose of this classification wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes, (2) the substrate is predominantly undrained hydric soil, and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year.* (Cowardin et al. 1979.)

Note that the definition used by the USFWS allows the use of a single parameter to determine if an area is a wetland. The definition also includes areas that may not be vegetated such as gravel bars and mudflats. The Corps of Engineers and Ecology definitions require the presence of all three parameters (vegetation, soil, and hydrology) for an area to be considered a wetland, and they both assume that wetlands generally are vegetated.

5.3.2 Biological vs. Regulated Wetlands

In some jurisdictions, all lands that meet the definition of "wetland" are regulated. However, it is not unusual for a jurisdiction to differentiate within its regulations between "wetlands" (i.e., biological wetlands) and "regulated wetlands" (i.e., wetlands that they intend to regulate). The definition of what constitutes a regulated wetland may vary from jurisdiction to jurisdiction.

In reviewing regulatory language from local wetland regulations, the three primary criteria used to differentiate between "wetland" and "regulated wetland" were:

- The category or rating of the wetland
- The size of the wetland
- Wetland types such as Prior Converted croplands and isolated wetlands

In general, a category or rating system has been historically used in regulatory language to differentiate between wetlands that need different degrees of protection. Rating systems are used by local jurisdictions to group wetlands based on physical characteristics and/or functions that the wetlands may provide and how those characteristics or functions are valued. Section 5.4 of this document describes the current state of the science on wetland ratings and the Washington State Wetland Rating System for eastern and western Washington.

The criterion of wetland size is usually a minimum below which the jurisdiction will not regulate a wetland. For example, the jurisdiction may allow no fill in wetlands larger than 10,000 square feet, or they may include language such as "Category 2 wetlands

larger than 0.25 acre cannot be altered." The historical rationale for the use of size as a regulatory criterion was the perception that "bigger is better" and the belief that small wetlands were less important and did not provide significant functions. The scientific literature of the last 10 years has made it clear that size does matter, but not in the way previously believed. Small wetlands have been shown in multiple studies to contain a significant diversity of plant and animal species. (See Section 5.3.3 for more information.)

Additionally, two other wetland types may be exempted from regulation: isolated wetlands and "Prior Converted" wetlands.

A U.S. Supreme Court decision in 2001 determined that most isolated wetlands are not subject to regulation under Section 404 of the federal Clean Water Act. However, the Court did not define "isolated," and the federal government has not issued any new guidance or regulations to clarify the situation. In general practice, the U.S. Army Corps of Engineers, the federal agency that administers the Clean Water Act, is considering isolated wetlands to be those of any size that have no direct surface water connection to any navigable waters. Washington State has determined that isolated wetlands are regulated by the Department of Ecology under the state Water Pollution Control Act (RCW 90.48). Since some local jurisdictions in Washington fashion their wetland regulations on the federal or state standards, it is important to consider the implications of not regulating isolated wetlands. Thus, scientific information on isolated wetlands is discussed in Section 5.3.4.

Prior Converted wetlands (or Prior Converted croplands) are another type of wetlands that are exempt from regulation by the federal government. Prior Converted wetlands are those wetlands that were drained or otherwise manipulated prior to December 23, 1985, for the production of commodity crops where inundation (ponding) does not occur for more than 14 consecutive days during the growing season. However, most of these areas still meet the three criteria for being a biological wetland. As with isolated wetlands, the Department of Ecology regulates Prior Converted wetlands under state law.

No information on areas meeting the definition of Prior Converted wetland (or cropland) was found in the scientific literature. However, many wetlands meeting these criteria would still be providing important functions given that the criteria for being designated "Prior Converted" require only that the wetland has been manipulated for production of commodity crops prior to 1985 and does not pond for more than 14 consecutive days during the growing season. If the agricultural activities were abandoned, the area could revert to a plant community characteristic of wetlands and, without maintenance of the hydrologic modifications, the wetland's water regime may revert to a condition more like that which existed prior to the alteration. Even if they have been maintained for agricultural production, many Prior Converted wetlands in western Washington pond during the winter and provide significant overwintering habitat for waterfowl (Zeigler, personal communication). Prior Converted wetlands may also provide important flood storage functions. Further analysis of the functions of areas designated as Prior Converted wetlands is needed.

No literature was found that discussed the ecological consequences of the legal bifurcation between biological wetlands and regulated wetlands. However, literature was found that discusses the functions and values provided by small wetlands and isolated wetlands, as discussed below.

5.3.3 Small Wetlands

The elimination of small wetlands is an issue that has gained increased attention over the past 10 years. Many regulations have preferentially allowed for filling of small wetlands because size is one of the most common characteristics used in determining wetland ratings at the local level. Smaller wetlands typically receive lower levels of protection. And yet, the loss of small wetlands is one of the most common cumulative impacts on wetlands and wildlife (Weller 1988, Tiner 2002).

In addition to the obvious loss of habitat for wildlife, fragmentation of habitat increases as small wetlands are removed, resulting in greater distances between wetland patches in the landscape. Semlitsch and Bodie (1998) found that creating greater distances between wetlands can have a significant effect on the ability of a landscape to support viable populations of amphibians, as juveniles dispersing from a source wetland may not be able to travel far enough to recolonize other surrounding (now distant) wetlands. Management priorities have focused on larger, semi-permanent wetlands, with the least emphasis on protecting the smaller, seasonal wetlands that are critical components of wetland complexes (Naugle et al. 2001).

This section describes studies of the use of small wetlands by wildlife and the role that small wetlands play in maintaining habitat connectivity. Studies of the relationship between wetland size and wildlife distribution have mostly focused on amphibians and birds. Few studies have examined how mammal use of wetlands relates to wetland size, and no studies of this relationship were found for macroinvertebrates or reptiles. No studies were found that documented the role that small wetlands play in providing water quality or hydrologic functions.

Small wetlands are differentiated only by their size. No definition of "small" is provided here because size is defined variously in scientific studies and by local regulatory or policy language. Small wetlands can have outlets, they can be in a floodplain, or they can be otherwise associated with a larger aquatic system.

Moler and Franz (1987) describe small wetlands as follows:

> *To a great extent, the unique values and functions of small, isolated wetlands have been overlooked. This oversight derives from several factors, perhaps foremost being the general tendency to think of small wetlands as being little more than subsets of larger wetlands. So long as the uniqueness of small wetlands is unrecognized, then it is intuitive to think of wetlands as declining in value directly as function of size. Similarly, so long as the unique values of isolated wetlands are unrecognized, it is understandable that connected wetlands might be*

considered of greater value. In reality, small isolated wetlands are biologically unique systems. Because of their isolation and small size, they support a very different assemblage of species than that found in larger, more permanently wet situations. The ephemeral nature of many small wetlands makes them unsuitable for species which require permanent water.

5.3.3.1 Amphibians and Small Wetlands

Snodgrass et al. (2000) undertook a study of amphibian use of wetlands to address three commonly held beliefs about small wetlands:

- They have short hydroperiods
- They support few species
- They support species that are also found in larger wetlands

Snodgrass et al. (2000) determined that amphibian species richness increases with length of hydroperiod. They also concluded that short-hydroperiod wetlands (smaller temporarily ponded wetlands) are also important in maintaining biological diversity in that they support species not found in larger wetlands with longer hydroperiods. The species they found in small wetlands were not a subset of those in larger wetlands but rather a unique group of species.

Similarly, amphibian richness in Puget Sound wetlands was found to have no correlation with wetland size. High richness occurred in some of the smallest wetlands (Richter and Azous 1995). The study indicates that small wetlands that are vegetatively simple can serve adequately as breeding habitats as long as favorable nonbreeding habitat is present nearby. Species richness also was not related to persistence of ponding.

Gibbs (1993) conducted a simulation model in Maine from which she theorized that small wetlands may be most important for wetland organisms with low population growth rates and low densities. The model demonstrated that the loss of small freshwater wetlands (less than approximately 5 acres [2 ha]) would result in a decline of total wetland area by 19% and total wetland number by 62%, while the average distance between wetlands would increase by 67% (Gibbs 1993). The model showed that the loss of small wetlands would result in a change (from 90% to 54%) of the area that would lie within the maximum migration distance of terrestrial-dwelling and aquatic-breeding amphibians. The risk of extinction would significantly increase for local populations of turtles, small birds, and small mammals that are currently stable even though the model showed no change in the risk of metapopulation extinction for salamanders or frogs. Amphibian populations in the study were buffered from the risk of extinction due to high rates of population increase. The model demonstrated that dispersal ability for amphibians is a predictor of population growth rate and density, not sensitivity of a population to loss of small wetlands.

5.3.3.2 Birds and Small Wetlands

Bird use of wetlands appears to have a stronger relationship to wetland size than that of amphibians. Bird richness was positively correlated with larger wetland size in a Puget Sound study of palustrine wetlands (Richter and Azous 2001b). This is attributed to the fact that larger wetlands in the study generally had greater structural complexity and a greater number of habitat types.

Martin-Yanny (1992) also found that bird species richness and abundance in wetlands of the Pacific Northwest are positively correlated with wetland size. However, they noted that habitat heterogeneity was a more important determining factor than wetland area in influencing bird species richness. Wetlands in highly urbanized watersheds had fewer neotropical migrant species, fewer ground-nesting birds, and more edge-tolerant (habitat generalist) species. This is because urbanizing watersheds tend to have smaller wetlands (less than 10 acres [4 ha]) with more edge habitat, making birds more susceptible to competition, predation, and nest parasitism. The author recommends preserving large wetlands or complexes of smaller wetlands that are connected by extensive upland buffers.

In northern prairie marshes, bird species richness was also seen to increase with marsh size and to decrease as the wetland became more isolated (Brown and Dinsmore 1986). Marshes that were part of wetland complexes showed higher species richness than isolated wetlands. Certain bird species used smaller marshes only when the marshes were part of a wetland complex. Large isolated marshes in the study often had lower species richness than smaller marshes that were part of wetland complexes. While bird species richness increased, the rate of increase slowed as the marshes became larger. In other words, they concluded that prairie marshes in the size range of 49 to 74 acres (20 to 30 ha) were more efficient in preserving bird species than larger marshes.

A study of agriculturally disturbed wetlands in western Oregon reached similar conclusions in finding that larger wetlands support more bird species (Budeau and Snow 1992). These authors also showed that wetlands of all sizes were important to water-birds.

However, in eastern Washington, Foster et al. (1984) found that waterfowl breeding use of wetlands in the Columbia Basin was greatest in smaller wetlands (less than 1 acre [0.4 ha]).

5.3.3.3 Mammals and Small Wetlands

The study that modeled the effects of the loss of small wetlands in Maine showed that local populations of small mammals faced a significant risk of extinction following the loss of small wetlands (Gibbs 1993). However, in a study of Puget Sound wetlands, Richter and Azous (2001c) concluded that wetland size alone was not a significant factor in determining mammal richness or abundance. They noted that small-mammal richness was most closely affected by the combined factors of:

- Wetland size
- Extent of retention of forest adjacent to the wetland
- Quantity of large woody debris within wetland buffers

5.3.4 Isolated Wetlands

Isolated wetlands are being addressed in this document because of the recent Supreme Court decision to exclude many isolated wetlands from federal regulation. This decision was made based on a legal interpretation of jurisdiction under the federal Clean Water Act. The key factor was the language in the Act that relates to navigable waters. The Court did not rule that isolated wetlands are less important than non-isolated wetlands, only that the intent of Congress in passing the Clean Water Act was to relate the protection of waters of the United States to navigability. The Court also did not provide any definition of what constitutes "isolation" for purposes of jurisdiction.

Much confusion has resulted from this decision and some statements have been made that isolated wetlands are less important or less worthy of protection. This section briefly summarizes some of the basic science on isolated wetlands, drawing heavily from the work of Tiner et al. (2002) who summarized much of the science on isolated wetlands. Readers are directed to this work for more detailed information. Additionally, the work of Hruby et al. (1999) in developing assessment methods for wetland functions in Washington provides important scientific information on depressional closed wetlands, a wetland type that constitutes the majority of isolated wetlands in Washington.

Wetlands can be defined as isolated based on their geographic isolation, ecological isolation, or hydrologic isolation (Tiner et al. 2002). For this discussion, isolated wetlands are defined by their hydrologic isolation—they do not have a surface outlet, even seasonally, to another water body. Although frequently described as closed depressions (Tiner et al. 2002), isolated wetlands can also be sloped wetlands where surface water, if present, re-enters the shallow groundwater zone at the base of the wetland and is not linked via surface flows to a downstream water body. Isolated wetlands are not necessarily small. They can be large systems with substantial heterogeneity and diverse habitat types (Tiner et al. 2002).

Generally, isolated wetlands provide most of the same functions as non-isolated wetlands and do so for the same reasons: position in the landscape, hydrologic regime, and type of soils and vegetation present. Basic functions of isolated wetlands as described by Hruby et al. (1999) and Tiner et al. (2002) are presented below:

- **Water quantity.** Isolated wetlands have no surface outlet. Therefore their ability to retain surface water may be significant, depending upon the surrounding topography. This provides potential flood storage because no surface water leaves the wetland to cause potential flooding downgradient.

- **Water quality.** Because they lack an outlet, isolated wetlands function as sediment traps when sediment moves into them. Isolated wetlands function as sinks for most dissolved and all sediment-associated nutrients and toxics because they have no outlets that allow materials to be transported downgradient. Their ability to take up and transform nutrients and toxics is similar to non-isolated wetlands and is largely a function of vegetation type and cover.

- **Wildlife habitat.** Isolated wetlands provide wildlife habitat functions similar to those of non-isolated wetlands, except in regard to habitat for migrating fish. The assessment models for general habitat suitability for depressional closed wetlands (isolated) and depressional outflow wetlands (non-isolated) are the same (Hruby et al. 1999). The habitat value of isolated wetlands is governed by the same factors as non-isolated wetlands (hydrologic regime, vegetation, connectivity to other habitats, etc.). Tiner et al. (2002) found that isolated wetlands provide essential habitat for a wide range of guilds and may be vital to maintaining viable, genetically diverse metapopulations. They state:

  From an ecological standpoint, isolated wetlands are among the country's most significant biological resources. In some areas, isolation has led to the evolution of endemic species vital for the conservation of biodiversity. In other cases, their isolation and sheer numbers in a given locality have made these wetlands crucial habitats for amphibian breeding and survival (e.g., woodland vernal pools and cypress domes) or for waterfowl and waterbird breeding (e.g., potholes). In arid and semi-arid regions, many isolated wetlands are veritable oases – watering places and habitats vital to many wildlife that use them for breeding, feeding, and resting, or for their primary residence.

5.3.5 Delineation Methods

In addition to the definition of what constitutes a wetland, the U.S. Army Corps of Engineers and Ecology have provided guidance on how to determine the edge of a wetland (i.e., how to delineate the wetland boundary). Delineating a wetland's boundary is a necessary step in the regulatory process because it factors into calculations of potential wetland impacts and determines the starting point for buffers and setbacks.

The U.S. Army Corps of Engineers published two federal manuals to delineate wetlands, one in 1987 and another in 1989. In subsequent years (1991, 1992, and with EPA in 1994) they released updates to clarify questions and provide regional guidance.

In the early 1990s there was substantial controversy over proposals to change the 1987 and 1989 federal delineation manuals. A substantial amount of literature was produced analyzing the effectiveness of the various delineation manuals for determining a wetland edge. In subsequent years, the use of the 1987 Federal Manual for Delineation of Wetland Areas has become the required legal standard for the U.S. Army Corps of Engineers.

As required by state legislation, Ecology issued the *Washington State Wetlands Identification and Delineation Manual* in 1996 (WAC 173-22-080, Ecology publication #96-94). Ecology's manual uses the original 1987 Corps of Engineers manual and incorporates changes in the manual made by the federal government since 1987. The state manual includes national guidance issued by the Corps in 1991 and 1992 (which is not used in the 1987 Corps manual), as well as regional guidance issued by the Corps and EPA in 1994. In addition, the state manual eliminated references and examples that were not relevant to Washington State and added examples and situations relevant to Washington. The 1996 state manual is required by statute (RCW 36.70A.175) to be used by local jurisdictions in implementing the Growth Management Act. Since the two manuals rely upon the same criteria and indicators for hydrology, soils and vegetation, proper use of either manual should result in the same boundary.

5.3.6 Summary of Key Points

- Regulatory agencies define the term "wetland" in slightly different ways.
- Local jurisdictions often differentiate between biological wetlands and regulated wetlands. The distinction is often based on the wetland rating and/or wetland size.
- The studies of the correlation of wetland size to wildlife use conflict somewhat in their findings, but most generally conclude that small wetlands are important habitats (particularly where adjacent buffer habitats are available) and that elimination of small wetlands can negatively impact local populations.
- Small wetlands provide habitat for a range of species that are not a subset of the species found in larger, more permanently inundated wetlands. For example, small wetlands do not just provide a smaller area for the same array of amphibian species that can be found in larger wetlands.
- Small wetlands are very important in reducing isolation among wetland habitat patches. Smaller wetlands provide significant habitat for wildlife and affect the habitat suitability of larger wetlands by reducing isolation on the landscape.
- The presence of small wetlands reduces the distance between wetlands and thus increases the probability of successful dispersal of organisms. This, in turn, likely increases the number of individuals dispersing among patches in a wetland mosaic, thereby reducing the chance of population extinction.
- Isolated wetlands provide the same range of wetland functions as non-isolated wetlands. Isolated wetlands provide important water quantity, water quality, and habitat functions.
- The U.S. Army Corps of Engineers 1987 wetland delineation manual and the 1996 *Washington State Wetlands Identification and Delineation Manual* are the

current standards to be used in determining the boundary of a wetland. Correct use of these two manuals should result in the same wetland boundary.

5.4 Wetland Rating Systems

Wetland rating systems (or categorizations) are one of the numerous procedures that have been developed to analyze wetlands, providing ways to identify, characterize, or measure wetland characteristics, functions, and social benefits (values). Ratings, as well as other procedures such as function assessment, are used by natural resource managers and regulators in a variety of contexts for regulating, planning, and managing the wetland resource (Bartoldus 1999). In the context of local regulations, rating systems are used to categorize wetlands based on different needs for protection. However, rating systems can often be used as one means to analyze wetlands.

Many different procedures to analyze wetlands have been developed in the last three decades. These range from detailed scientific evaluations that may require many years to complete, to the judgments of individual experts during one visit to a wetland. For example, Bartoldus (1999) summarized 40 different tools that had been developed up to 1998 and that are used to meet the needs of regulating and managing wetlands.

Although many different rating-type tools have been developed, the literature search for this document did not uncover any analyses of the effectiveness of rating systems at protecting the wetland resource. It is assumed that better protection for wetlands is provided with improved understanding of wetland functions and values (e.g., Roth et al. 1993, National Research Council 1995).

Scientific rigor is often time consuming and costly. For regulatory use, tools are needed to provide some information on the functions and values of wetlands in a time- and cost-effective way. One way to accomplish this is with an analytical tool that categorize wetlands by their important attributes or characteristics based on the collective judgment of regional experts. Categorization methods, such as rating systems, are relatively rapid but can still provide some scientific rigor (Hruby 1999).

The rapid method most commonly used for analyzing wetlands in eastern and western Washington has been the *Washington State Wetland Rating System* (Ecology 1991, 1993, Hruby 2003). This rating system or some modification of it has been incorporated in the wetland regulations of at least 20 counties in the state and many cities and towns as well (Office of Community Development, personal communications).

In the first edition of the *Washington State Wetland Rating System*, the term "rating" was not used in a manner that is consistent with its definition in the dictionary, and this has caused some confusion. The method does not rate the wetlands and generate a relative estimate of value (e.g., high, medium, low). Rather, it is a categorization of wetlands based on specific criteria such as sensitivity to disturbance and rarity in the landscape.

The rating system was designed to differentiate between wetlands based on their sensitivity to disturbance, their significance, their rarity, our ability to replace them, and the functions they provide. However, this rating system is not intended to replace a full assessment that may be necessary to determine the levels of performance for numerous functions or plan and monitor a compensatory mitigation project. As noted in the Washington State rating system:

> *The rating categories are intended to be used as the basis for developing standards for protecting and managing the wetlands to minimize further loss of their resource value. The management decisions that can be made based on the rating include the width of buffers necessary to protect the wetland from adjacent development, the ratios needed to compensate for impacts to the wetland, and permitted uses in the wetland* (Hruby 2003).

The rating system for both eastern and western Washington is being revised by Ecology in conjunction with teams of wetland experts and local planners in each region who are providing technical input and field testing. The rating system for eastern Washington will be finalized in fall 2003 and the western Washington rating system will be finalized by early 2004. The goal of the revisions is to reflect the best and most current science on wetlands and how they function (using three broad groups of functions—hydrologic, water quality, and habitat) while maintaining rapidity and ease of use. You can access the draft rating system for eastern Washington at the following web site: www.ecology.wa.gov/programs/sea/wetlan.html

Wetland rating systems used in other parts of the nation

Categorization systems have also been used in other parts of the United States to manage wetlands. Other states have wetland categorizations as part of their wetland laws and rules, and other jurisdictions have used them to help manage wetlands for specific projects. For example:

Vermont adopted a law (10 VSA Chapter 37, Section (a) (7-9)) mandating that rules be adopted to identify Vermont's significant wetlands. The rules categorize wetlands into three classes of which the first two are considered "significant" (Vermont Department of Environmental Conservation 1999).

New Jersey has a wetland categorization included directly in its law (NJAC 7:7A). Criteria are provided for categorizing wetlands into (1) freshwater wetlands of exceptional resource value, (2) wetlands of ordinary resource value, and (3) wetlands of intermediate resource value.

New York has adopted rules that categorize wetlands into four categories based on ecological associations, hydrologic features, pollution control features, cover types, and distribution and location (6 NYCRR Part 664.5).

West Eugene, Oregon developed a method for a plan based on "needs for protection" (City of Eugene 2000).

North Carolina created a GIS-based system that characterizes the "significance" of wetlands based on several landscape and function-based criteria (Gainey and Roise 1998).

5.4.1 Other Characteristics Used for Rating

Some freshwater wetlands in Washington are categorized in the Washington State Wetland Rating System based on important characteristics that are not specifically related to functions. These characteristics include rarity on the landscape, sensitivity to disturbance, and difficulty in restoring or creating such wetlands through mitigation efforts (Ecology 1991, Hruby 2003). At present these wetland types have been redefined for eastern Washington (Hruby 2003) and include:

- Bogs
- Alkali wetlands
- Mature and old-growth forested wetlands
- Vernal pools
- Wetlands identified by the Washington State Department of Natural Resources as "Natural Heritage wetlands"

Each of these types is described in more detail below. The list will be expanded as the wetland rating system for western Washington is revised. Currently, the 1993 edition of the rating system for western Washington identifies the following freshwater wetlands as ones with special characteristics: bogs, mature forested wetlands, and Natural Heritage wetlands.

5.4.1.1 Bogs

Many of the scientific studies of bogs have been published in Europe and the northern parts of the United States such as Minnesota and Maine. There has not been extensive research on bogs in Washington State. This summary of the literature is not intended to be a thorough synthesis but provides basic background information regarding characteristics of bogs requiring special consideration for management.

Predominance of Organic Soils

Bogs are peatlands (wetlands with organic soils) that have been classified according to their shape, chemistry, plant species, and vegetation structure (Gore 1983). The common factor in bogs is the presence of organic soils or peat, which result from the accumulation of poorly decomposed plant material. The optimum conditions for peat formation occur in cool, humid climates in a location with poorly drained soil.

The rate of peat accumulation is generally quite low, although it can vary with site-specific factors. Heathewaite and Gottlich (1993) report rates of accumulation ranging from 2 to 4 inches (5 to 10 cm) every 100 years. Durno (1961) lists a range of 0.5 to 4.3 inches (1.2 to 11 cm) accumulation every 100 years. In Washington, Rigg (1958) reports peat accumulation of 1 inch (2.5 cm) in 40 years for the west side of the Cascades and 1 inch in 50 years on the east side. Peat can be as little as 8 inches (20 cm) deep to over 45 feet (15 m) deep (Heathewaite and Gottlich 1993).

The three ways that peat is formed, described below, illustrate the lengthy process of peat and bog formation and help explain why bogs are almost impossible to recreate through compensatory mitigation (see below and in Chapter 6).

- In a “filled-lake” sequence, open water progresses to a sedge or moss community that gradually builds a mat over the water, evolving into a bog, bog forest, and then climax community (Conway 1949).
- “Paludification” occurs when bogs invade the surrounding forest. Sphagnum species cause a rise in the water table as peat layers compress and impede drainage (Heathewaite and Gottlich 1993).
- A “flow-through succession” occurs when surface flows are modified. Organic matter builds up to the point where surface flows are diverted around the peat mound. As it builds, the mound becomes isolated from groundwater, relying solely on precipitation as its water source (Klinger 1996).

Studies have shown, on the other hand, that many bogs remain very stable for thousands of years as a sphagnum moss/shrub community, even though succession to a forested community can occur (Klinger 1996).

Acidity and Poor Nutrients

Bogs have unusual hydrodynamics and chemistry for wetlands. They typically only receive precipitation and very localized surface runoff as their sources of water. As a result, many essential nutrients such as nitrogen occur in low concentrations. The upper layers of peat, formed by slowly decomposing sphagnum, are often strongly acidic, usually with a pH of 4 or less.

Bogs typically support plant species that are specially adapted to these harsh growing conditions. Sphagnum moss, as well as other mosses, usually dominate the system. Ericaceous shrubs such as Labrador tea (*Ledum gladulosum*) are also common.

Trees can grow in bogs but at a very slow rate due to the poor growing conditions. In studies in the Pacific Northwest, Rigg (1918) found tree growth in sphagnum peat soils was slow. Rigg determined that hemlock (*Tsuga heterophylla*) grew in sphagnum soils at a rate that was only 27% of its growth rate in productive upland soils, and that Douglas-fir (*Pseudotsuga menziesii*) grew in sphagnum at only 16% of its growth rate in upland soils. He measured the annual growth of western red cedar (*Thuja plicata*) as only 0.02 inches (0.6 mm).

Although persistent wet conditions, low soil oxygen, and high acidity are important factors, it is actually the lack of available nutrients, or the inability of plants to absorb nutrients because of acidity (Moore and Bellamy 1974), that most influences the flora of bogs. Most bog species have developed special adaptations to these conditions and out-compete more common wetland plants (Mitsch and Gosselink 2000). Therefore, this makes bog species susceptible to nutrient loading and changes in acidity (as well as alterations in water source that can precipitate these changes) that would enable other species to establish and dominate.

Bogs in Western and Eastern Washington

In western Washington, Kunze (1994) has characterized numerous types of peatlands, including bogs and fens. She identified 10 types of sphagnum bog communities in the Puget Trough region and 14 in the Olympic Peninsula/southwest Washington. They occur in the lowlands of the Puget Trough in depressions, oxbows, and old lake beds. These typically have a raised center with a moat around the edge. Bogs and fens also occur on the Olympic Peninsula and in southwest Washington where they can occupy basins, slopes, and flat to rolling ground, as well as forming along low-gradient streams. Bogs in the foothills of the Cascades include sloping bogs, which are transitional between sphagnum bogs and fens.

Peatlands in eastern Washington have not been classified to the extent of those in western Washington. However, 50 peatlands were identified by Rigg (1958). Forty-four of those identified were located in the northeastern corner of the state. They included fens

associated with flowing water, bogs formed in depressions, and some along lake margins. Six peat systems were found in scabland channels and depressions on the Columbia Plateau.

Difficulty in Restoring Bogs

Researchers in Northern Europe and Canada have found that restoring bogs is difficult, specifically in regard to plant communities (Bolscher 1995, Grosvermier et al. 1995, Schouwenaars 1995, Schrautzer et al. 1996), water regime (Grootjans and van Diggelen 1995, Schouwenaars 1995), and/or water chemistry (Wind-Mulder and Vitt 2000). In fact, restoration may be impossible because of changes to the biotic and abiotic properties (Shouwenaars 1995, Schrautzer et al. 1996).

It is apparent that true restoration of a raised bog ecosystem is a long-term process. In *Restoration of Temperate Wetlands*, Joosten (1995) (in Roos) states:

> *Long term studies in bog regeneration indicate that restoration of bogs as self-regulating landscapes after severe anthropogenic damage is impossible within human time perspective, because the necessary massive re-establishment of bog key species and renewed accumulation of peat require centuries.*

Refer to Chapter 6 for more information on the challenges in restoring bogs.

5.4.1.2 Alkali Wetlands

Alkali wetlands are characterized by the occurrence of non-tidal, shallow saline water. In eastern Washington these wetlands contain surface water with specific conductance (a measure of salinity) that exceeds 3,000 micromhos per centimeter. These wetlands provide the primary habitat for several species of migratory shorebirds and are also heavily used by migrating waterfowl. They also have unique plants and animals that are not found anywhere else in eastern Washington. For example, the small alkali bee that is used to pollinate alfalfa and onion for seed production lives in alkali systems. This bee is a valuable natural resource for agriculture in the western United States and especially in eastern Washington (Deplane and Mayer 2000). The “regular” bees which pollinate fruits and vegetables are generally too large to pollinate the small flowers of these commercially important plants.

The salt concentrations in alkali wetlands have resulted from a relatively long-term process of groundwater surfacing and evaporating. These conditions cannot be easily reproduced through compensatory mitigation because the balance of salts, evaporation, and water inflows is hard to reproduce, and no references were found suggesting this has ever been attempted. Alkali wetlands are also rare in the landscape of eastern Washington. Of several hundred wetlands that were surveyed and visited by wetland scientists during field work for the state’s function assessment methods and the rating system for eastern Washington (Hruby et al. 2000, Hruby 2003), only nine could be classified as alkali.

5.4.1.3 Mature and Old-Growth Forested Wetlands

No mature or old-growth forested wetlands have ever been successfully created or restored through compensatory mitigation. A mature forested wetland may require 80 years or more to develop, and the full range of functions performed by these wetlands may take even longer (Stanturf et al. 2001). The actual time required to reconstruct old-growth forests and their soil properties (in contrast to mature forests) is unknown (Zedler and Callaway 2000). These forested wetlands provide important functions associated with wetlands as well as habitat functions associated with mature and old-growth forests. (Washington State Department of Fish and Wildlife 1999).

5.4.1.4 Vernal Pools

Vernal pool wetlands occur in eastern Washington and are formed when small depressions in bedrock or in shallow soils fill with snowmelt or spring rains. They retain water until the late spring when reduced precipitation and increased evapotranspiration lead to a complete drying out. The wetlands hold water long enough throughout the year to allow some strictly aquatic organisms to flourish, but not long enough for the development of a typical wetland environment (Zedler 1987). Vernal pools often contain upland species during the summer after they dry out and may be difficult to identify as jurisdictional wetlands during part of the year.

Vernal pools in the scablands are the first to melt in the early spring. This open water provides areas where migrating waterfowl can find food while other, larger bodies of water are still frozen. Furthermore, the open water provides areas for pair bonding of waterfowl (Friesz, personal communications). Thus, vernal pools in a landscape with other wetlands provide a critical habitat function for waterfowl (Hruby 2003).

5.4.1.5 Natural Heritage Wetlands

Natural Heritage wetlands are those that have been identified by scientists of the Washington State Natural Heritage Program as high-quality, relatively undisturbed wetlands, or wetlands that support state threatened or endangered plant species.

The Natural Heritage Program has identified important natural plant communities and species that are very sensitive to disturbance or threatened by human activities and maintains a database of these sites. The program's web site states:

> *Some natural systems and species will survive in Washington only if we give them special attention. By focusing on species at risk and maintaining the diversity of natural ecosystems and native species, we can help assure our state's continued environmental and economic health.* (Washington Department of Natural Resources no date.)

5.4.2 Summary of Key Points

- Wetland rating systems provide a rapid method to identify, characterize, categorize, or estimate relative wetland functions and values. This information is used in regulating and managing wetlands.
- The rapid method most commonly used for analyzing wetlands in eastern and western Washington has been the *Washington State Wetland Rating System*. The rating system was designed to differentiate between wetlands based on a broad grouping of functions that they provide (hydrologic, water quality, and habitat) as well as other characteristics (listed in the next bullet). However, this rating system does not replace a full assessment of wetland functions that may be necessary to determine the level of performance for specific functions or plan and monitor a compensatory mitigation project.
- In the rating system, some wetlands are categorized because of their rarity on the landscape, sensitivity to disturbance, or difficulty in restoring or creating such wetlands through mitigation efforts, not because of the functions these wetlands perform. The wetland types in Washington that are included in the rating system because they have these other characteristics include bogs, alkali wetlands, mature and old-growth forested wetlands, vernal pools, and Natural Heritage wetlands.

5.5 Buffers

Buffers are another common element of wetland regulations. Buffers are vegetated areas adjacent to an aquatic resource that can, through various physical, chemical, and/or biological processes, reduce impacts from adjacent land uses. Buffers also provide the terrestrial habitats necessary for wildlife that use wetlands to meet their life-history needs. In this document we collectively call these processes that buffers provide the "functions" of buffers. Buffers and other adjacent upland areas provide habitat for other wildlife species that do not commonly use wetlands. This document does not address those functions of upland habitats.

The primary purpose of buffers is to protect and maintain the wide variety of functions and values provided by wetlands (or other aquatic areas). The physical characteristics of buffers—slope, soils, vegetation, and width—determine how well buffers reduce the adverse impacts of human development and provide the habitat needed by wildlife species that use wetlands. These characteristics are discussed in detail in this section.

The subject of buffers is well documented in the scientific literature. The research on buffers has occurred worldwide, and this section includes literature from a variety of regions when it was found to be relevant. In particular a variety of literature related to agricultural practices and vegetated filter strips from the north-central United States and south-central Canada is relevant to some agricultural practices in Washington State, especially east of the Cascades. In addition, studies on buffers in urban and suburban settings conducted in the Pacific Northwest region are clearly relevant.

The majority of research on buffers tends to focus on the processes that buffers provide to filter sediment or take up nutrients (i.e., their influence on water quality). Far fewer studies look at the influence of a buffer's physical characteristics on attenuating surface water flow rates, except as it relates to water quality. The long-term effectiveness of buffers in providing such mechanical and biological processes is not well documented in the literature and may represent a critical need for future research.

The literature on buffers related to wildlife is, in general, less focused. Most studies document the needs of a particular species or guild relative to distances for breeding or other life-history needs within a radius from aquatic habitats. There is substantial literature on the implications of habitat fragmentation and connectivity, some of it related specifically to agricultural practices, forestry practices, or the impacts of urbanization. This literature does not specifically address the role of buffers in providing connectivity between wetlands and other parts of the landscape. The reader is referred to Section 4.11 in Chapter 4, which discussed the effects of habitat loss and fragmentation.

Older research studied the tolerance limits of wetland wildlife for disturbance—how closely a disturbance can approach animals before they are flushed from wetlands—with particular emphasis on waterfowl. These studies tend to be older than 1990 and focus on the prairie pothole region of North America. Where the findings are germane and where they have not been superceded by more recent work, they are included.

In addition to papers on specific research studies, multiple compilations and syntheses of "buffer" literature have been completed since 1990. Synthesis papers were compiled by Castelle and other authors (1992, 1994, 2000) and another was compiled by McMillan (2000) as a master's thesis. These compilations include literature that was published prior to 1990, but much of the work they rely on is considered seminal to the effectiveness of buffers in protecting wetlands and contributing to habitat. Therefore these synthesis documents are used in this report as direct sources when no more recent research was found to supercede the earlier findings.

This section also cites literature related to stream buffers and riparian areas when the findings are relevant to the influence these areas have on the adjacent aquatic resource. The literature on stream buffers related to microclimate, water quality influences, and some habitat characteristics is particularly relevant because the ways buffers protect and maintain these functions is similar whether they are adjacent to streams or wetlands.

5.5.1 Terms Used to Describe Buffers

The scientific literature varies widely on the terms used to denote the area that serves to reduce impacts to wetlands from adjacent land uses and provide habitat for parts of the life-cycle of many species. Common terms include:

- Buffer
- Wetland setback

- Vegetated filter strip
- Buffer strip
- Riparian area
- Riparian zone
- Riparian corridor

These terms can be differentiated as those that are a product of regulations or policy language, and those that define or describe an ecological condition or location (Castelle et al. 1994). Terms such as "buffer," "wetland setback," or "vegetated filter strip" are most commonly applied in an administrative context to denote the landscape immediately adjacent to an aquatic resource, the dimensions of which are legally determined. The terms "buffer strip" or "vegetated filter strip" may imply a relatively undisturbed, vegetated area that helps attenuate the adverse effects of land uses adjacent to a wetland. For example, Norman (1996) provides this definition:

> *Buffer strips are strips of vegetated land composed in many cases of natural ecotonal and upland plant communities which separate development from environmentally sensitive areas and lessen these adverse impacts of human disturbance.*

The terms "riparian areas" or "riparian zones" are defined by many to denote ecologically discernable ecotones (transition zones) along aquatic resources where the presence or action of surface waters, or the presence and duration of shallow groundwater, influences the structure and composition of the vegetation community (Lowrance et al. 1995, Harper and MacDonald 2001). The term "riparian corridor" is defined by Naiman et al. (1993) as "encompass(ing) the stream channel and that portion of the terrestrial landscape from the high water mark towards the uplands where vegetation may be influenced by elevated water tables or flooding, and by the ability of the soils to hold water."

5.5.2 Functions Provided by Buffers

The literature is broadly consistent on the ways in which buffers can provide for the protection and maintenance of wetland functions. These include:

- Removing sediment
- Removing excess nutrients (phosphorous and nitrogen)
- Removing toxics (bacteria, metals, pesticides)
- Influencing the microclimate
- Maintaining adjacent habitat critical for the life needs of many species that use wetlands

- Screening adjacent disturbances (noise, light, etc.)
- Maintaining habitat connectivity

As noted by Castelle and Johnson (2000), buffers can be both ecological sources and sinks. They can control or limit the effects of land uses upslope of the aquatic resource (act as a sink), and they can contribute biological benefits to the aquatic resource (act as a source). Naimen et al. (1992) summarize the range of functions provided by buffers along streams as follows:

> *It is well known that riparian vegetation regulates light and temperature regimes, provides nourishment to aquatic as well as terrestrial biota, acts as a source of large woody debris,…regulates the flow of water and nutrients from uplands to the stream, and maintains biodiversity by providing an unusually diverse array of habitat and ecological services.*

These same functions can be attributed to wetland buffers (Castelle et al. 1992, Desbonnet et al. 1994, McMillan 2000).

The literature also describes the physical, chemical, and/or biological characteristics of a buffer that determine the functions it provides. The most frequently cited physical characteristics that influence the effectiveness of a buffer are:

- Landscape position of the buffer
- Vegetation characteristics (composition, density, and roughness—for example, downed material)
- Percent slope
- Soils
- Buffer width and length (adjacent to the source of impacts)

Only two of the physical characteristics noted above can be easily managed (vegetation characteristics and buffer width/length), while the others are characteristics that do not lend themselves to manipulation.

By far the issue of greatest interest with respect to buffers is the question of how wide a buffer needs to be in order to be effective in protecting a wetland (or other aquatic resource). While the literature is unanimous that buffers provide important functions that protect wetlands and provide essential habitat for many species, there is wide-ranging discussion about how much buffer is necessary to be effective in providing a particular level of function (Young 1980, Booth 1991, Castelle et al. 1994, Norman 1996, Dosskey 2000, McMillan 2000, Rickerl 2000).

For ease of discussion as to the effective widths of buffers, the functions of buffers listed above are grouped into two major categories:

- Water quality (discussed in Section 5.5.3)
- Wildlife habitat (discussed in Section 5.5.4)

Buffers and their influence on wetland hydroperiod, as described in the few studies found on this subject, are summarized in the shaded box on the next page.

The following literature sources are generally consistent in describing what functions buffers provide to aquatic resources as well as the physical parameters that influence a buffer's ability to provide these functions: Budd et al. (1987), Phillips (1989), Castelle et al. (1992, 1994), Naiman et al. (1992), Belt and O'Laughlin (1994), Desbonnet et al. (1994), Norman (1996), Dillaha and Inamdar (1997), Dosskey (2000), Van der Kamp and Hayashi (1998), Liquori (2000), McMillan (2000), Todd (2000), Townsend and Robinson (2001), Dosskey (2001).

Buffers alone have limited influence on wetland hydroperiod

As described in detail in Chapter 3, human land uses such as agricultural practices, clearing, and land development alter the movement and storage of surface water and groundwater within a wetland's contributing basin. These changes can significantly affect the hydroperiod of wetlands and other aquatic resources, causing an adverse effect on many wetland functions (Azous and Horner 2001). There is little published literature on the effectiveness of buffers in ameliorating the effect of changes in land use within the contributing basin on wetland hydroperiod. Some of the literature determined that wetland buffers are far less effective at maintaining wetland hydroperiod than other mechanisms such as controlling impervious surfaces and utilizing effective stormwater management practices (Herson-Jones et al. 1995).

Research in the Puget Sound Basin has agreed that changes in the land cover type in the contributing basin have a stronger influence on the resulting hydroperiod of the wetland than the buffer does (Booth 1991, Azous and Horner 2001). An exception may be for wetlands that have a very small contributing basin. However, the rate and manner in which stormwater enters the wetland following land use changes in the contributing basin will most often shift from sheet flow and interflow to one or more point sources, resulting in a potential change in hydroperiod. Based on hydroperiod models using the HSPF model (Hydrologic Simulation Program Fortran, U.S. EPA) for areas west of the Cascades, the wetland will tend to receive more water more quickly in the fall and will receive less water for a shorter period in the spring, resulting in a shift in the seasonal hydroperiod.

Buffer width is usually not sufficient to counteract the influence of land use changes and stormwater management facilities within the wetland's contributing basin.

5.5.3 Buffers and Protection of Water Quality

Buffers protect the water quality of wetlands through four basic mechanisms:

- They remove sediment (and attached pollutants) from surface water flowing across the buffer
- They biologically "treat" surface and shallow groundwater through plant uptake or by biological conversion of nutrients and bacteria into less harmful forms
- They bind dissolved pollutants by adsorption onto clay and humus particles in the soil
- They help maintain the water temperatures in the wetland through shading and blocking wind

Literature describing the different ways that buffers maintain and improve water quality in wetlands and other aquatic areas is abundant. There is also considerable research on the effective widths that provide a relative percentage of removal of sediments, nutrients, and some toxics emanating from various sources. Four categories of water quality improvement are discussed below:

- Removing sediment
- Removing nutrients
- Removing toxics and pathogens
- Maintaining microclimate

For each of these categories, a summary is provided on what the literature says about the relationship between buffer width or other characteristics and the buffer's effectiveness in providing that type of water quality improvement. A summary table is included that lists the range of buffer widths for each category and the literature references that substantiate those findings.

5.5.3.1 Removing Sediment

Characteristics that Influence a Buffer's Ability to Remove Sediment

A buffer's ability to remove sediment from surface water flows depends upon several physical characteristics of the buffer. Sediment removal occurs when (Castelle et al. 1992, Dillaha and Inamdar 1997, Phillips 1989):

- Flows are slowed sufficiently to allow particles to fall out
- Physical filtering by vegetation and roots mechanically removes sediments from the water column
- The slope of the buffer is of a low enough gradient to preclude formation of rills and scouring
- There is large woody debris on the ground to create roughness
- The infiltration rate of the soils allows water to move through the soils rather than on the surface

The way sediment-laden water enters a buffer influences the ability of the buffer to slow the flows sufficiently to allow sediment deposition. Several studies noted that vegetated buffers are only effective at removing sediments if sediment-laden waters enter the buffer as sheet flow, rather than in channels or rivulets (Phillips 1989, Booth 1991, Castelle et al. 1992, Desbonnet 1994, Belt and O'Laughlin 1994, Sheridan et al. 1999). Norman (1996) cites work conducted by Schueler in 1987 that found buffers in urban settings were most effective at removing sediments where slopes were less than 5% and waters entered the buffer in shallow, dispersed sheet flow. Norman surmised that, "The rate of

removal of pollutants appears to be a function of the width, slope, and soil permeability of the (buffer) strip, the size of the contributing runoff area, and the runoff velocity."

In other research, Sheridan et al. (1999) found that the greatest reduction in sediment loading occurs in the initial "treatment" stages using a vegetated filter strip that is managed and mowed. Their research found the greatest removal of sediments (56 to 72%) and reduction in flow rates occurs in the outer portion of a vegetated filter strip (the strip closest to the source of sediment). Grass filter strips provided removal ranging from 78 to 83% of suspended sediments.

The ability of a buffer to provide physical filtering of sediments also depends on the condition of the vegetation and the surface roughness. Belt and O'Laughlin (1994) noted that when vegetation, rocks, or other obstructions were eliminated from the buffer surface, sediment-laden waters flowed further into (or through) a buffer. Buffers were found to be effective in removing sediments only if flows were shallow and broad, not narrow and incised. The presence of woody debris and vegetative obstructions on the ground surface (roughness) was found to slow flows, inhibit the formation of rills, and facilitate sediment deposition.

In contrast, hydrologic models created by Phillips (1989) estimated that surface roughness would be of minor concern and buffer width was not critical as long as a minimum 49-foot (15 m) buffer was maintained. This study was based on estimated models, whereas Belt and O'Laughlin's work was based on field measurements.

Phillips (1989) also emphasized the importance of slope. He states, "Results show that where solid-phase pollutants transported as suspended or bed-load in overland flow are the major concern, slope gradient is the most critical factor, followed by soil hydraulic conductivity." Slope gradient is critical because on slopes greater than 5% sheet flow can start to become channelized. Channelized flows have faster rates, more erosive powers, and less contact with vegetation (Norman 1996). Faster moving water has the capacity to carry fine sediment particles farther than slower flows, even moving through dense vegetation.

In his research in urbanizing settings, Booth (1991) notes that buffers adjacent to aquatic resources may have limited ability to filter and slow flows caused by stormwater. He found (1) in some instances the buffers no longer existed in a natural vegetated condition, or (2) once development occurred and the buffer was subdivided into multiple private ownerships, maintaining an intact buffer was not possible, or (3) the increased volumes and rates of flows were too significant to be controlled by conditions within a vegetated buffer.

Buffers were found to facilitate reduction of sediment from active agricultural fields in several studies:

- Welsch (1991) found that a three-tiered buffer system on a shallow slope, with the first tier (closest to the source of sediment) composed of dense herbaceous vegetation, maximized sediment removal. (See Section 5.5.6 for a discussion of the three-tiered system.)

- Dosskey (2001) noted in agricultural settings that vegetated buffers retain pollutants by reducing the flow rates and filtering surface runoff from fields.
- Assessing management options to control non-point-source pollution (sediment, nitrogen, and phosphorus) in agricultural settings, Yocom et al. (1989) recommended the use of vegetated filter strips between actively cropped land and adjacent wetlands.

Buffer Width and Effectiveness in Removing Sediment

As noted above, the ability of a buffer to remove sediment is based on the condition of the buffer and its slope as well as the characteristics of the incoming sediment. The following variables all contribute to the sediment removal effectiveness of a buffer:

- The velocity of sediment transport (in surface water)
- The size of sediment particles from the source materials
- The density of the vegetation present
- The presence and extent of large woody debris
- Surface roughness within the buffer

However, the relationship between the width of the buffer and its effectiveness is non-linear. The largest particles and the greatest percentage of particles are dropped in the outer portions of the buffer (closest to the source of sediment). In these outer areas, the rate of surface flow begins to diminish as the water is slowed by vegetation and woody debris. Slower water movement allows particles to drop out of the water column.

In 1982, Wong and McCuen derived a formula to model a buffer's ability to remove sediments based on sediment particle size, the slope within the buffer, the rate of surface runoff, and the amount of vegetation and woody debris (roughness) in the buffer (Castelle et al. 1994). The model predicted that there would be a point of relative diminishing returns for function vs. width. For example, "If the sediment removal design criteria were increased from 90 to 95% on a 2% slope, then the buffer widths would have to be doubled from 30.5 to 61 m (100 to 200 ft)." In other words, the model predicted that the width of the buffer would have to double to achieve an additional 5% removal of sediment after 90% of it had already been removed from the water column. Desbonnet et al. (1994) determined that a small buffer (7 feet [2 m]) could effectively remove up to 60% of suspended sediment, while a buffer of up to 82 feet (25 m) would be needed to remove 80%.

These findings are consistent with others who have found that progressively larger buffer dimensions are required to filter out finer particles (Norman 1996). These and other studies are summarized in Table 5-1.

See Section 5.5.5 for discussion of the ability of buffers to continue providing sediment removal over the long term.

Table 5-1. Summary of studies on sediment control provided by buffers of various widths.

Author(s)	Date	Buffer Width	Comments
Broderson	1973	200 feet (61 m)	Effective sediment control "even on steep slopes" as cited by Castelle and Johnson (2000)
Desbonnet et al.	1994	6.6 – 82 feet (2 – 25 m)	60% removal in 6.6 feet (2 m); 80% removal required 80 feet (25 m)
Desbonnet et al.	1994	16 – 49 feet (5 – 15 m)	On grassy buffers on slopes with less than 5% slope, removed all but the finest particles. Cited by McMillan (2000)
Ghaffarzadeh et al.	1992	16 – 49 feet (5 – 15 m)	Found 85% removal in 30-foot (9.1 m) buffers as cited by Castelle and Johnson (2000)
Horner and Mar	1982	200 feet (61 m)	80% of sediments. As cited by Castelle and Johnson (2000)
Lynch, Corbett, and Mussallem	1985	98 feet (30 m)	75 to 80% removal of sediment from logging activities into wetlands. As cited by Castelle and Johnson (2000)
Norman	1996	9.8 feet (3 m): sands 49.9 feet (15.2 m): silts 400 feet (122 m): clays	Distances required for effective removal of progressively smaller particle sizes
Wong and McCuen	1982	100 – 200 feet (30.5 – 61 m)	90% at 100 feet (30 m), need 200 feet (61 m) to obtain 95% removal effectiveness. Cited by Castelle et al. (1994)
Young	1980	80 feet (24.4 m)	92% sediment removal rate from feedlot through vegetated buffer strip. Cited by Castelle et al. 1994

5.5.3.2 Removing Nutrients

Characteristics that Influence a Buffer's Ability to Remove Nutrients

Nutrients are transported into wetlands via sediment-laden water or dissolved in surface or shallow subsurface flows. The primary nutrients of concern are nitrogen and phosphorous. Buffers remove nitrogen and phosphorous through a variety of mechanisms that are similar to the mechanisms present within the wetland itself, as described in Chapter 2.

As much as 85% of phosphorous in surface waters is bound to sediments (Karr and Schlosser 1977) and thus can be removed via sediment removal in buffers. Phosphorus and other nutrients may be effectively reduced in surface waters by filtering and uptake; however, dissolved forms of nitrogen are not affected by surface processes and can be more effectively removed in the buffer through subsurface contact with fine roots

(Muscutt et al. 1993, Townsend and Robinson 2001). Lowrance et al. (1995) confirm that the areas where improvements in water quality are the most effective are where precipitation moves across, through, or near the rooting zone of a forested buffer. These findings are similar to those of Phillips (1989), who found that longer contact of dissolved pollutants through wider vegetated buffers was the most important factor for effective removal.

Buffer Width and Effectiveness in Removing Nutrients

It is difficult to compare studies of buffer width and effectiveness at removing nutrients because the basic parameters of the studies differ greatly. Some studies were conducted in field settings while others occurred in experimentally designed plots. There were differences in the loading rate of nutrients, the types of soils, and the vegetation in the buffers. Some studies examined only nitrogen or phosphorous removal, whereas others combined different nutrients. The result is that reported effectiveness of buffer widths for removing nutrients ranges from a few meters to hundreds of meters. Studies are listed in Table 5-2.

In a synthesis of research on nitrogen removal, McMillan (2000) found nitrogen can be effectively removed in buffer strips ranging from 20 to 98 feet (6 to 30 m) wide. He cites work by two research groups (Patty et al. 1997, Daniels and Gilliam 1996) that 47 to 99% removal of nitrogen can be achieved in buffers ranging from 20 to 66 feet (6 to 20 m) wide. This is not totally consistent with synthesis results presented by Desbonnet et al. (1994) that "well configured" buffers (with ideal slope, soils, and vegetation) as small as 30 feet (9 m) could reduce as much as 60% of nitrogen, while 197-foot (60 m) buffers would be necessary for 80% nitrogen removal.

The literature also describes a range of buffer widths necessary for phosphorus removal. Studies of buffers as small as 13 feet (4 m) wide and as large as 279 feet (85 m) wide found phosphorus removal rates of 50% to over 90% (see Table 5-2).

Overall, a consistent pattern emerges from the literature. The largest relative percent removal of phosphorus occurs within the outer portions of the buffer (closest to the source), while larger buffers are required to remove increasingly more of the nutrients. This consistent finding substantiates the conclusions of many that initial contact causes sediment-associated nutrients to be deposited, while dissolved nutrients require longer residence time and prolonged contact with vegetation for effective uptake (removal from the water column) to occur.

Castelle and Johnson (2000) surmised in their literature review that nutrient removal may have a similar non-linear relationship to buffer width as sediment removal. However, Phillips (1989) found that buffer width was a more critical element for dissolved nutrients (especially nitrogen), because wider buffers provided more prolonged contact with the rooting zone and time for uptake and conversion. Phillips did not report widths of buffers related to a certain percent of removal or effectiveness.

Limited research has been done on the long-term effectiveness of buffers for nutrient removal when there is an ongoing nutrient source present on the outside edge of the buffer. See Section 5.5.5 for a discussion.

Table 5-2. Summary of studies on nutrient removal provided by buffers of various widths.

Author(s)	Date	Width	Comments
Daniels and Gilliam	1996	20 – 66 feet (6 – 20 m)	47-99% removal of nitrogen, cited by McMillan (2000)
Desbonnet et al.	1994	30 feet (9 m): 60% removal 197 feet (60 m): 80% removal	Small buffers could have effective removal rates for nitrogen; much larger buffers are necessary for a significant increase in effectiveness
Desbonnet et al.	1994	Averages: 39 feet (12 m): 60% 279 feet (85 m): 80%	When all the findings from the literature synthesis were averaged, the average removal efficiencies were non-linear: larger buffers were needed for increases in effectiveness
Dillaha	1993	15 feet (4.6 m): 70% 30 feet (9.1 m): 84 %	Percent removal of suspended solids and their associated nutrients with vegetated filter strips. As cited in Todd (2000)
Dillaha	1993	15 feet (4.6 m): 61 % 30 feet (9.1 m): 79 %	Removal of phosphorus with vegetated filter strips. As cited by Todd (2000)
Dillaha	1993	15 feet (4.6 m): 54% 30 feet (9.1 m): 73%	Removal of nitrogen with vegetated filter strips. As cited by Todd (20000)
Doyle, Stanton and Wolf	1977	12.5 feet (3.8 m) forested 13.1 feet (4 m) grass	Reduced nitrogen, phosphorus, and potassium levels. Cited by Castelle and Johnson (2000), McMillan (2000)
Edwards et al.	1983	98 feet (30 m)	50% removal rate of phosphorus. As cited by McMillan (2000)
Lowrance et al.	1992	23 feet (7 m)	Forested buffer zones were effective at removing nitrate through plant uptake and microbial denitrification
Lynch, Corbett and Mussallem	1985	98 feet (30 m)	Forested buffers reduced soluble nutrient levels from logging activities to "appropriate" levels. Cited by Castelle and Johnson (2000)
Patty et al.	1997	20 – 66 feet (6 – 20 m)	47 - 99% removal of nitrogen, as cited by McMillan (2000)
Peterjohn and Correll	1984	164 feet (50 m)	Forested buffer strips provided "dramatic reductions in nutrient loads from crops" as cited by Belt and O'Laughlin (1994)

Author(s)	Date	Width	Comments
Shisler, Jordan, and Wargo	1987	62 feet (19 m)	Forested riparian buffers effectively removed up to 80% and 89% of phosphorus and nitrogen, respectively. Cited by Castelle and Johnson (2000)
Thomson et al.	1978	39 – 118 feet (12 – 36 m)	Found a range of removal effectiveness of 44 to 70%. As cited by McMillan (2000)
Vanderholm and Dickey	1978	> 853 feet (260 m)	Removal of 80% of nutrients, solids, and BOD from feedlot runoff with shallow (<0.5%) buffer slopes. Cited in Castelle et al. (1998)
Young et al.	1980	69 feet (21 m): 67% removal 89 feet (27 m): 88% removal	Removal of phosphorus, as cited by McMillan (2000)
Xu, Gillam, and Daniels	1992	33 feet (10 m)	Significant reductions in nitrate through a mixed herbaceous and forested buffer strip. As cited by Castelle and Johnson (2000)

5.5.3.3 Removing Toxics and Pathogens

Characteristics that Influence a Buffer's Ability to Remove Toxics and Pathogens

A buffer's ability to remove toxicants and pathogens is one of the least thoroughly studied. At this time, it represents a significant data gap. Castelle and Johnson (2000) note the lack of research on pathogens such as fecal coliform bacteria and toxicants such as pesticides. Many of the studies they examined are quite old, but little recent research was found to supplement these older studies. Therefore, the conclusions presented from the synthesis of the previous work are provided here.

Gilliam (1994) also confirms in his work that little to no research is available on the effective removal of fecal coliforms or various pesticides. Much of the work assessed the effectiveness of removal of nutrients and toxics, without identifying a dimension of width necessary to provide that removal.

Toxics (pesticides and metals) can be removed by buffers through sedimentation, biological uptake by vegetation, adsorption onto clay or humus particles in the soil of the buffer, or degradation of the toxics through biochemical processes (McMillan 2000, Patty et al. 1997).

As mentioned in the discussion of sediment removal, Welsch (1991) described the use of a three-tier buffering system for the most effective removal of sediments and their associated toxics. The outermost tier (closest to the source of impacts) was a densely vegetated filter strip, managed to ensure no erosion or rill formation. He found the most effective removal of sediments and the toxics adhered to sediment particles was through surface sheet flows through the vegetated filter strip. The middle tier was subject to some

management activities (limited agriculture or limited tree harvest), while the innermost tier was undisturbed natural vegetation. Dissolved nutrients and some toxics were not affected by physical filtering unless there was prolonged contact with the rooting zone through the shallow groundwater table. See Section 5.5.6 for further discussion.

Castelle and Johnson (2000) note that the apparent effectiveness of small buffers in removing toxics is due to the adsorption of many toxics to sediment particles. When vegetated buffers are effective at filtering sediments, they will also be effective at filtering those toxics and nutrients adhered to the sediments.

One study in Saskatchewan (Donald et al. 1999) found that the concentrations of agricultural pesticides and herbicides in wetlands were influenced by the timing of precipitation relative to the applications of the chemicals. They noted that buffer width may influence exposure of the wetland to these chemicals, but they did not quantify what buffer widths related to the effectiveness of removing chemicals.

Buffer Width and Effectiveness in Removing Toxicants and Pathogens

Table 5-3 summarizes studies on the effectiveness of toxicant and pathogen removal provided by buffers of various widths.

Table 5-3. Summary of studies on pathogen control provided by buffers of various widths.

Author(s)	Date	Width	Comments
Doyle, Stanton and Wolf	1977	12.5-foot (3.8 m) forested buffers 13.1-foot (4 m) grass buffers	Reduction in fecal coliform bacteria levels as cited by Castelle and Johnson (2000)
Grismer	1981	98-foot (30 m) grass filter strip	Removal of 60% of fecal coliform bacteria as cited by McMillan (2000)
Young et al.	1980	115-foot (35 m) grass buffer	Reduced microorganisms to acceptable levels. Cited by McMillan (2000)

5.5.3.4 Maintaining Microclimate

The influence of buffers on microclimate is most often thought of in the context of shading for maintaining water temperature. This is well documented in the literature in relation to the effects on streams (Johnson and Stypula 1993, Belt and O'Laughlin 1994, Castelle and Johnson 2000, McMillan 2000). In those documents, literature focused on streams and their buffers is almost exclusively relied upon to discuss the influences of buffers on water temperature. No literature was found that specifically examined the influence of buffers on the water temperatures and microclimates within wetlands.

Although it may be tempting to deduce that the benefit of forested shade in moderating water temperatures is the same in wetlands as in streams, it is not reasonable to apply findings on the dimensions of stream buffers for shading directly to wetlands. There are

too many variables related to differences in water budgets (e.g., the relative influence of groundwater on a seasonal basis, whether the wetland has an inlet/outlet, etc.). Also, the physical configurations of a large open-water wetland, a small fully vegetated wetland, and a linear stream corridor may not provide reasonable parallels. With these limitations in mind, some relevant findings are provided below.

Forests can create shade and also block the wind, which can help moderate temperatures in adjacent aquatic systems (McMillan 2000). Stable water temperature helps maintain water quality because cooler water can carry higher loads of dissolved oxygen, which is important for many aquatic biota. Warmer water can also result in a looser bond between sediment particles and nutrients, which could result in an increase in nutrient loading in warmer aquatic systems (Karr and Schlosser 1977).

Microclimate influences can also extend from large wetlands into the adjacent forests. Harper and MacDonald (2001) conducted research on boreal forests near lakes and found a "distinct lake edge community" of about 131 feet (40 m) width. The lake edge community tended to have greater structural diversity, less canopy cover, fewer snags, greater amounts of coarse woody debris, and greater number of saplings and mid-canopy trees than the interior forest. Changes in the distribution of vegetation species were along a shade tolerance gradient, but the authors postulated that moisture gradient or water table depth also had an influence. Their research was conducted within forests adjacent to open water lakes, but it would be valid to extrapolate their findings to forested communities adjacent to permanent, large open wetlands that would create the same "light and shade" effect. The findings imply that large open aquatic systems influence the adjoining upland community for over 120 feet (37 m) distance into the interior of the forested buffer. Thus buffers not only influence temperatures and wind effects in a wetland but research identifies that large aquatic systems may have a reverse positive influence on the vegetation structure and species diversity of the buffer. This can thereby affect some of the habitat discussed later in this chapter.

Table 5-4. Summary of studies on the influence of microclimate provided by buffers of various widths.

Author(s)	Date	Width	Comments
Harper and MacDonald	2001	Approx. 131 feet (40 m)	Influence of large aquatic systems on adjacent upland forest composition and structural complexity

5.5.3.5 Summary of Key Points

- The use of buffers to protect and maintain water quality in wetlands (removing sediments, nutrients, and toxicants) is best accomplished by ensuring sheet flow across a well vegetated buffer with a flat slope (less than 5%).
- Significant reductions in some pollutants, especially coarse sediments and the pollutants adhered to them, can be accomplished in a relatively narrow buffer of

16 to 66 feet (5 to 20 m), but removal of fine sediments requires substantially wider buffers of 66 to 328 feet (20 to 100 m).

- Removal of dissolved nutrients requires long retention times (dense vegetation and/or very low slope) and, more importantly, contact with fine roots in the upper soil profile (i.e., soils that are permeable and not compacted). Distances for dissolved nutrient removal are quite variable, ranging in the literature from approximately 16 to 131 feet (5 to 40 m).
- The literature is consistent in finding that it takes a proportionally larger buffer to remove significantly more pollutants because coarse sediments and the pollutants associated with them drop out in the initial (outer) portions of a buffer. It takes a longer time for settling, filtering, and contact with biologically active root zones to remove fine particles and dissolved nutrients.
- The role of buffers in protecting the microclimate of streams is well documented and may be applicable to wetlands, but no specific data on buffers and wetland microclimate maintenance were found.

5.5.4 Buffers and Wildlife Habitat

Wetland buffers are essential to maintaining viable wildlife habitat because they perform three overlapping functions:

- Buffers can provide an ecologically rich and diverse transition zone between aquatic and terrestrial habitats, including necessary terrestrial habitats for many wildlife species that use wetlands
- Buffers can screen wetland habitat from the disturbances of adjacent human development
- Buffers may provide connectivity between otherwise isolated habitat areas

In regard to wildlife, most of the scientific research is not directly focused on the effectiveness of buffers for maintaining individuals or populations of species that use wetlands. Some of the research simply documents use of upland habitats adjacent to wetlands by wildlife to meet their life-history needs. For example, a substantial body of research identifies the distances that amphibians may be found away from a wetland edge, but what is not well documented are the implications to amphibian populations by providing buffers that are smaller than those identified ranges.

The following discussion summarizes the literature on buffers related to wildlife that use wetlands for the three essential functions listed above. Several documents are cited that represent a synthesis of scientific literature on the effectiveness of buffers for protecting wildlife-related functions of wetlands. Even though these documents include some research conducted prior to 1990, they have been included where relevant.

There is substantial literature on the implications to wildlife populations from fragmenting habitats as a result of human activities. However, this research was not necessarily conducted to address the effectiveness of various buffer widths. The literature on this topic is mentioned because of the management implications for the long-term viability of wetland-dependent species. The reader is referred to Section 4.11 in Chapter 4 for a detailed discussion of habitat connectivity and fragmentation.

5.5.4.1 Maintaining Terrestrial Habitat Adjacent to Wetlands

Buffers provide a transition between aquatic and terrestrial environments and are a critical component of the habitat of wildlife that use wetlands. The specific habitat functions provided by wetland buffers include:

- Sites for wildlife for foraging, breeding, and nesting
- Cover for escape from predators or adverse weather
- Source of organic matter that provides habitat structure and food
- Areas for dispersal and migration related to both individuals and populations (buffers may connect or be part of corridors)

As defined previously, buffers are predominantly upland habitat communities that lie adjacent to aquatic habitats. They are a different habitat type than the wetland and their presence increases habitat heterogeneity by providing niches for more species. First described by Leopold (1933) as the "edge effect," this phenomenon features higher use of transition zones by wildlife, particularly between aquatic and terrestrial habitats. It has been demonstrated in studies of birds (Beecher 1942, McElveen 1977), mammals (Bider 1968), and amphibians (Bury 1988). The same pattern has been demonstrated in the Pacific Northwest in studies by Oakley et al. (1985), Knight (1988), and Cross (1988). Research conducted in the Puget Sound lowlands confirmed the relationship between habitat diversity and species richness when the greatest species richness of birds and small mammals was found within the first 1,640 feet (500 m) adjacent to a wetland boundary (Richter and Azous 2001b, 2001c).

Protection of upland areas adjacent to wetlands is critical to helping ensure that wetland-dependent wildlife populations have access to the habitat features necessary to meet their survival requirements. Wetland-dependent species, such as many amphibians, aquatic invertebrates, waterfowl, and some mammals, require access to wetlands for critical stages of their life-history. Many more species use wetlands, as well as other aquatic systems such as streams, lakes, or rivers, to meet various life-history needs. Research shows that species that were assumed to be dependent upon wetlands also depend upon adequate and appropriate upland habitats to maintain viable populations (Foster et al. 1984, Bury 1988, Washington Department of Wildlife in Castelle et al. 1992).

In addition, vegetated buffers protect habitat in wetlands by maintaining the microclimate (through temperature moderation), as discussed previously, and by providing a source of organic matter to aquatic systems. This includes both large organic debris (logs, root

wads, limbs), which provides habitat structure in aquatic environments, and particulate and dissolved organic matter, which provides a source of food for invertebrates (Brown 1985, Groffman et al. 1991).

In coastal wetlands in South Carolina, Braccia and Batzer (2001) found that large woody debris within wetlands was critical for both aquatic and terrestrial invertebrate populations. They identified that the source of the large woody debris within the wetlands was from the adjacent uplands. The forest conditions in adjacent uplands therefore can have a significant influence on wetland biota because the aquatic invertebrates form the foundation of many food chains in aquatic settings (Castelle et al. 1994).

Buffer Width and Effectiveness in Protecting Wetland Habitat and Providing Habitat in Adjacent Uplands

This section summarizes the literature that identified ranges of widths of uplands that protect wetland habitat and/or that provide adjacent upland habitat for wildlife species that use wetlands. The literature presents findings in a variety of ways. Some studies identify the distance that target species range from a wetland source, while other researchers identified the distances that species travel between wetlands. Synthesis documents outlined recommendations for buffer widths based on a review of research findings. Some of the literature identified use of habitats by broad categories of wildlife guilds, while other studies focused on limited guilds or even individual species.

It is important to understand that the range of buffer widths identified and discussed in the literature is a reflection of many variables including the objectives of the research, the species/guilds studied and their varied life-history needs, and the methods of the research. Thus, it is not appropriate to choose a single study or buffer dimension to justify a buffer dimension, whether large or small. It is critical to incorporate the life-history requirements of the range of targeted species when considering buffer dimensions. Synthesis documents clarify that a range of upland habitat buffer dimensions may be appropriate depending upon site considerations, landscape context, and targeted species.

For example, in summarizing the literature he reviewed on buffer effectiveness, McMillan (2000) concluded, “An appropriate buffer to maintain wildlife habitat functions for all but the most highly degraded wetlands would be comprised of native tree and shrub vegetation and range from 30 to 100 meters [98 to 328 feet].” Other authors have reached similar conclusions, with their buffer recommendations varying depending on the type of wildlife, life-history stage, intensity of adjacent land use, and surrounding landscape (Groffman et al. 1991, Castelle et al. 1992, Desbonnet et al. 1994, Semlitsch 1998). Because there is often substantial information on the needs for some specific wildlife groups, the research findings that are relevant for birds, amphibians, reptiles, and mammals are provided below. Following this discussion, Table 5-5 provides a summary of literature on general habitat needs in relation to buffer sizes.

One element that was not found for this synthesis was the implications of the condition of the upland buffer relative to its provision of wildlife habitat. In several studies on the use

of upland buffers by native species, the study identified that the buffer was upland forest. However, no studies were reviewed for this synthesis that compared wildlife use of mature forested buffers with buffers composed of meadow, shrubland, logged forest, or younger forests. Some research has identified the importance of intact forest habitat to wetland-related species (Azous and Horner 2001, Richter 1997), but a comparison study was not found for this synthesis.

Birds

The research on birds ranges from studies in individual species to summaries on bird species richness. A tremendous amount of research on waterfowl exists, with the majority being conducted in the prairie pothole region of the United States. This section focuses on studies or syntheses that are relevant to the Pacific Northwest.

The Puget Sound Stormwater Management Research Program found that a distance of 1,640 feet (500 m) from a wetland edge was necessary to account for total species richness of birds (Richter and Azous 2001b). In a study of bird use of freshwater wetlands in urban King County, Washington, Milligan (1985) determined that bird species diversity was strongly correlated with the percentage of the wetland boundary that was buffered by at least 49 feet (15 m) of trees and shrubs.

In eastern Washington, Foster et al. (1984) determined that 68% of waterfowl nests were in upland areas within 98 feet (30 m) of the wetland edge, whereas it would take a 312-foot (95 m) buffer to encompass 95% of the nesting sites.

Temple and Cary (1988) created a computer model whose results may relate to the breeding success of forest birds using wetland buffers. Estimating the effects of habitat fragmentation on birds breeding in the interior of forests in Wisconsin, their model predicted that nesting success was strongly correlated to distance to the edge of a forest. The computer model predicted a success rate of 70% for nests greater than 656 feet (200 m) from the forest edge, 58% for a distance of 328 to 656 feet (100 to 200 m), and only 18% for nests less than 328 feet (100 m) from the forest edge. Applying these findings to wetland buffers, those less than 100 feet (30 m) in width might not be expected to support bird species that nest in forest interiors. The authors concluded that without “recruits” (birds moving into appropriate habitat niches from farther afield), the continued fragmentation of forest habitats could lead to local extinction of populations of birds that use the interior of forests.

Amphibians

The research on amphibians and buffers in relation to their habitat needs comes both from studies in the Pacific Northwest and literature summaries from around the United States. Findings are rather consistent in that amphibians range substantial distances from breeding locations in a wetland to fulfill their life-history needs. On the west side of the Cascades there appears to be a preference for forested habitats adjacent to breeding sites, and urban uses near breeding sites seem to have a negative influence on amphibian abundance.

Detailed findings include:

- A study in the Puget Sound lowlands documented a decline in amphibian richness in wetlands where forest in the contributing watershed was diminishing; results were not linked to buffer dimensions (Richter and Azous 2001a).
- In a study in King County by Ostergaard (2000), the greatest use of stormwater ponds by native breeding amphibians was found when 3,280 feet (1,000 m) of forested habitat was available adjacent to the pond.
- A study of pond-breeding salamanders found that a buffer of 534 feet (164 m) would be needed to encompass 95% of adult and juvenile salamanders; this buffer range may apply to other similarly mobile species (Semlitsch 1998). Buffers of 98 to 328 feet (30 to 100 m) were recommended along riparian zones depending upon slope, stream width, and adjacent use (Semlitsch 1998).
- Salamanders use upland habitats over 1,969 feet (600 m) from the edge of wetlands for non-breeding life-history stages. Sustaining viable wetland-dependent amphibian species requires maintaining the connection between wetlands and terrestrial habitats. (Semlitsch 1998).

See Table 5-5 for further information on these studies.

In addition, Knutson et al. (1999) found a positive correlation between the presence of forest around the perimeter of the wetland and amphibian abundance, and a negative correlation to urban land uses on the perimeter.

Reptiles

Research on reptiles and buffers in the Pacific Northwest was not found. Research on freshwater turtles in North Carolina found that turtles used a wide area for nesting and terrestrial hibernation in uplands surrounding the ponds where breeding occurred (Burke and Gibbons 1995). They found that a 902-foot (275 m) buffer was required to protect 100% of the nest and hibernation sites. Protecting 90% of the sites required a 240-foot (73 m) buffer. The authors concluded that most buffer requirements are inadequate to protect turtle habitat for all stages of their life-history.

Mammals

Use of wetlands by mammals depends upon adjacent uplands. The literature indicates that even a wetland-dependent mammal such as a beaver uses upland habitats an average of 100 feet (30 m) from the wetland edge in eastern Washington and over 300 feet (100 m) distant in western Washington (Castelle et al. 1992). Research on small mammals found the greatest concentration of species near riparian corridors, with some species found within that riparian corridor that were not found farther away in upland habitats (Cross 1985).

Dimensions of effective buffers for mammals are more difficult to discern from the literature because they depend upon the species' life-history. Also, as discussed in Section 4.11 of Chapter 4, habitat linkages and fragmentation may be more critical for the sustainability of some populations.

As part of the Puget Sound Stormwater Management Research Program, Richter and Azous (2001c) found that the highest richness of small mammals was in wetlands with at least 60% of the first 1,640 feet (500 m) of buffer in forest cover. Other findings of this program include:

- The preservation of large woody debris within the wetland and adjacent upland forest is important for maintaining small-mammal habitat.
- Small-mammal richness was best associated with the combined factors of wetland size, adjacent forest, and the quantity of large, coarse woody debris within the wetland and its buffer.
- In southwestern Oregon, Cross (1985) conducted research on small mammals in "leave-strips" adjacent to streams within zones of forest that had been harvested. He found that the richness of small-mammal species was highest in the riparian zone closest to the stream, intermediate in the transition zone, and lowest in the upland zone. (The zones were defined by vegetation composition, not by dimension.) Because riparian habitats provide more niches for species, it is expected that such habitats would maintain greater species richness (Cross 1985).

Cross also found no species in the upland zone that were not found in the riparian zone, but he found five species present in the riparian zone that were not present in the upland or transition zones. A strip averaging 220 feet (67 m) wide supports mammal communities at similar numbers and richness to the nearby undisturbed riparian corridor. This study focused on small mammals which, relative to large mammals, have small home ranges. Therefore, the study is not broadly applicable to appropriate leave-strip dimensions for larger species.

Table 5-5 presents a summary of literature on wildlife and buffer/upland habitat use that was relevant to this synthesis. As noted previously, some of the research is specific to individual species, some is focused on a particular guild or group of similar species, some looks at life-history patterns (nesting distances), and other sources represent synthesis documents of buffer effectiveness. These distances do not necessarily reflect the literature relative to human disturbance and/or habitat fragmentation, which are discussed in the next sections.

It is difficult to synthesize the findings of the research on wildlife and the width of buffers into simple generalizations that can be readily applied. When looking at life-history needs (nesting sites, foraging ranges, etc.) the distances presented in the literature range from 98 feet (30 m) (Foster et al. 1984, Castelle et al. 1992) to 3,280 feet (1,000 m) (Richter 1997). These are distances measured in the field that species ranged, nested, or foraged from a wetland edge.

Other authors have presented their own synthesis or recommendations of effective buffer ranges based on review of the literature. These range from 49 feet (15 m) (Desbonnet et al. 1994) to 328 feet (100 m) (Groffman et al. 1991, Castelle et al. 1992, Desbonnet et al. 1994, McMillan 2000). Note that Desbonnet et al. (1994) recommend a range of buffer dimensions based on site conditions, species of interest, and proposed adjacent land uses; hence their studies are cited at both ends of the distance spectrum.

Table 5-5. Summary of studies on wildlife habitat provided by buffers.

Author(s)	Date	Width	Comments
Allen	1982	328 – 590 feet (100 – 180 m)	Mink use: generally concentrated within 330 feet (100 m) of water but will use upland habitats up to 590 feet (180 m) distant
Burke and Gibbons	1995	240 feet (73 m): 90% 902 feet (275 m): 100%	Buffer to encompass % nesting and hibernation of turtles in North Carolina
Castelle et al.	1992	197 – 295 feet (60 – 90 m): Western Washington 98 – 197 feet (30 – 60 m): Eastern Washington	Range for all species they noted. Range for all species they noted.
Castelle et al.	1992	263 feet (80 m) avg. - 590 feet (180 m)	Wood duck nesting locations from wetland edge (non-Washington data)
Castelle et al.	1992	98 feet (30 m): Eastern Washington 328 feet (100 m): Western Washington	Distance of beaver use of upland habitats from water edge
Chase et al.	1995	98 feet (30 m) or more	100 feet (30 m) would be "adequate"; buffers larger than 100 feet needed to meet habitat needs, including breeding for birds and some mammals
Cross	1985	220 feet (67 m)	Forested "leave-strips" for small mammal richness adjacent to streams in SW Oregon
Desbonnet et al.	1994	49 – 98 feet (15 – 30 m): low intensity 98 – 328 feet (30 – 100 m): high intensity	Variable buffer widths using adjacent land uses as decision-making criteria
Fischer et al.	2000	98 feet (30 m) minimum	Literature review; majority of literature cited recommends buffer widths of 330 feet (100 m) for reptiles, amphibians, birds, and mammals

Author(s)	Date	Width	Comments
Foster et al.	1984	98 feet (30 m): 68% of nests) 312 feet (95 m): 95% of nests	Waterfowl breeding use of wetlands in the Columbia Basin greatest in smaller (<1 acre [0.4 ha]) wetlands; 68% of waterfowl nests within 100 feet (30 m) of wetland edge; to encompass 95% of waterfowl nests would require 310 feet (95 m) of buffer
Groffman et al.	1991	197 - 328 feet (60 - 100 m)	For most wildlife needs
Groffman et al.	1991	328 feet (100 m)	Neotropical migratory bird species
Howard and Allen	1989	197 feet (60 m)	For most wildlife needs
McMillan	2000	98 – 328 feet (30 – 100 m)	Based on a synthesis of literature
Milligan	1985	49 feet (15 m)	Bird species diversity strongly correlated with the percentage of the wetland boundary buffered by at least 50 feet (15 m) of tree and shrub vegetation
Norman	1996	164 feet (50 m)	To protect wetland functions; more buffer may be required for "sensitive wildlife species"
Ostergaard	2001	3,280 feet (1,000 m)	Forested habitat surrounding stormwater ponds, related to native amphibian richness
Richter	1996	3,280 feet (1,000 m)	Literature review and synthesis
Richter	1996	3,280 feet (1,000 m)	Native amphibian use
Richter and Azous	2001b	1,680 feet (512 m)	Distance from wetland edge necessary to include all bird richness in Puget Sound lowland wetlands
Richter and Azous	2001c	1,640 feet (500 m): 60%	Highest small-mammal richness when 60% of first 1,640 feet (500 m) of buffer was forest habitat
Semlitsch	1998	1,969 feet (600 m)	Salamanders
Semlitsch	1998	228 – 411 feet (69.6 - 125.3 m) 539 feet (164.3 m) for 95% of all species	Six species of adult salamanders and two species of juveniles; mean distance from wetland edge was 228 feet (juveniles) – 411 feet (adults). To incorporate 95% of all species, buffer mean would have to be 539 feet
Short and Cooper	1985	164 – 328 feet (50 – 100 m)	164 feet (50 m) for foraging
Temple and Cary	1988	> 656 feet (200 m): 70% success 328 – 656 feet (100 – 200 m): 58% success < 328 feet (100 m): 18% success	Nesting success rates for interior-dwelling forest birds related to distance into the interior of a forest from the forest edge

5.5.4.2 Screening Adjacent Disturbances

Wetland buffers screen wildlife from human activities. Disturbance from humans can come in the form of noise and light (indirect effects) or from human presence/movement (direct effects). Noise and light can disrupt feeding, breeding, and sleeping habits of wildlife. Many wildlife species in wetlands are disturbed by unscreened human activity within 200 feet (61 m) (Washington Department of Wildlife in Castelle et al. 1992). Dense shrubs and trees in a wetland buffer can limit intrusion and screen out noise, light, and movement from adjacent human development (Castelle et al. 1992).

In addition, domestic pets such as dogs and cats can adversely affect wetland wildlife by preying on some wildlife species and are particularly damaging to ground-nesting species (Churcher 1989). See Section 4.12.5 in Chapter 4 for further discussion.

The effect of noise on wildlife is a topic of growing concern. Little research exists on the effective buffer widths required to filter sounds for wildlife. See Section 4.12.3 in Chapter 4 for a discussion of current literature on the effects of noise on wildlife.

Groffman et al. (1991) determined that 105 feet (32 m) of dense forested buffer was necessary to reduce noise from commercial areas to background noise levels. Shisler et al. (1987) differentiated between the impacts of low-intensity land uses (agricultural, recreational, low-density housing) and high-intensity land uses (high-density residential, commercial/industrial). They found that low-intensity land uses could be effectively screened with vegetated buffers of 49 to 98 feet (15 to 30 m), while high-intensity land uses required buffers of 98 to 164 feet (30 to 50 m).

Direct sighting of humans approaching was found to disrupt birds (change their behavior or cause flushing) between 46 and 164 feet (14 to 50 m) (Shisler 1987, Josselyn et al. 1989, Rodgers and Smith 1997). Looking specifically at great blue herons, Short and Cooper (1985) documented that they would flush from their nests if humans approached within 328 feet (100 m). Buffers between 46 and 164 feet (14 to 50 m) may be required to screen wildlife from direct observation of humans, while larger buffers (328 feet or 100 m) were documented as necessary to screen nesting herons.

Other researchers differentiated between the types of activities humans are engaged in and their effects on wildlife. Humans walking toward birds were studied to see how closely they could approach before birds flushed from perches or stopped foraging. In Florida, Rodgers and Smith (1997) found that humans could approach 46 to 112 feet (14 to 34 m) before flushing, but automobiles flushed birds at 61 to 78 feet (18.5 to 24 m). Interestingly they found that bird-watching (as opposed to humans who were simply walking) had the greatest adverse impacts on birds. They surmised this was due to the human behavior of stopping and standing with binoculars at one point for a prolonged time.

Cooke (in Castelle et al. 1992) analyzed 21 wetland sites in western Washington and concluded that buffers smaller than 50 feet (15 m) were generally ineffective in screening human disturbance from alterations such as noise, debris, and altered use of the buffer.

Table 5-6 summarizes the findings of the literature related to the disturbance limits or screening effects of a buffer for various wildlife species.

Table 5-6. Summary of studies on screening provided by buffers.

Author(s)	Date	Width	Comments
Castelle et al.	1992	200 feet (61 m)	General wildlife considerations
Cooke	1992	50 feet (15 m)	Analyzed 21 sites in King County. Buffers less than 50 feet were often disturbed by human activities and were not effective at screening "human effects". Cited in Castelle et al. (1992)
Groffman et al.	1991	105 feet (32 m)	Dense forest to filter sound from commercial land uses to natural background levels
Josselyn et al.	1989	49 – 164 feet (15 – 50 m)	Unscreened human activity within 50 – 164 feet was disruptive to waterbirds in San Francisco Bay area
Rodgers and Smith	1997	46 to 112 feet (14 –34 m) 61 to 78 feet (18.5 – 24 m)	Waterbirds in Florida: flushing distance from walkers 46 – 112 feet; flushing distance from autos 61 – 78 feet. Nature observation had greatest impact of human walking activities. Nesting birds tolerated closer human approach than birds that were perching/foraging
Shisler et al.	1987	50 - 100 feet (15 – 30 m) 100 – 164 feet (30 – 50 m)	Low-intensity land uses (agriculture, recreation, and low density residential): 50 - 100 feet High-density residential housing and commercial/industrial: 100 - 164 feet Most effective buffers had steep slopes, dense shrubs
Short and Cooper	1985	328 feet (100 m)	328 feet to buffer nesting great blue herons from human disturbance

5.5.4.3 Maintaining Habitat Connections

Increased isolation of wetlands and fragmentation of habitats result directly from human conversion of habitats to other uses. Buffers can play a role in reducing habitat fragmentation by serving as upland habitat directly adjacent to a wetland, or by providing an area that can connect or be part of a corridor that connects wetlands with upland habitats or other water bodies. Generally, buffers, as applied in a regulatory context, are rarely designed to provide these connections. Typical buffer widths generally are insufficient to link wetlands to other habitats. In addition, maintaining linkages from one habitat type to another on distinct parcels is often not a consideration when properties are reviewed case by case.

In general, the literature states that for terrestrial species with wide-ranging habits, connectivity between breeding, feeding, and refuge sites is critical for maintaining population viability. One may assume that this applies only to large terrestrial mammals. However, research has shown that many native amphibians on the west side of the Cascades can range 3,280 feet (1,000 m) from source wetlands into other wetlands or surrounding upland habitats (Richter 1997). Ostergaard (2001) found the greatest amphibian richness in sites that had upland forest habitat surrounding the site by 3,280 feet (1,000 m). Richter and Azous (2001b) found that a radius of 1,680 feet (512 m) surrounding a wetland was necessary to include all the bird richness of species utilizing the source wetland.

See Section 4.11 in Chapter 4 for further discussion of habitat connectivity and fragmentation.

5.5.4.4 Summary of Key Points

- There is no simple generalized answer for what constitutes an effective buffer width for wildlife considerations. The width of the buffer is dependent upon the species in question and its life-history needs, whether the goal is to maintain connectivity of habitats across a landscape, or whether one is simply trying to screen wildlife from human interactions.
- The majority of wildlife species in Washington use wetland habitats for some portion of their life-history needs. Many wetland-dependent species (those that depend upon wetlands for breeding, brood-raising, or feeding) depend upon surrounding upland habitats as well for some life-history stages.
- Many terrestrial species that are dependent upon wetlands have broad-ranging habits, some over 3,280 feet (1,000 m) from the source wetland. Although this might be expected for large mammals such as deer or black bears, it is also true for smaller species such as salamanders and other amphibians.
- Human access and land uses adjacent to wetlands influence the use and habits of wildlife through visual and auditory intrusions, as well as elimination or degradation of appropriate upland habitats. Even “passive” activities such as bird/nature-watching have been shown to have effects on roosting and foraging birds.
- Synthesis documents that evaluated many studies of wetland buffers for habitat protection recommend buffers widths of 50 to 300 feet (15 to 100 m) for most situations.

5.5.5 Buffer Maintenance and Effectiveness over Time

Buffers provide various functions to protect wetlands for as long as the buffers themselves remain intact. Buffer areas can be altered over time by human disturbance

and natural events such as windstorms. In addition, some researchers have raised the issue of whether buffers have a long-term carrying capacity to improve water quality with regard to filtration and binding of pollutants.

5.5.5.1 Human Alteration to Buffers

Human activities are the most common mechanism for altering buffers over time. Buffer functions can be reduced if vegetation is cut or trampled, soils are compacted, sediment loading surpasses the filtering capability of the vegetation, or surface water flows create channels and subsequent erosion.

Cooke (in Castelle et al. 1992) analyzed 21 wetland sites in western Washington and concluded that buffers less than 50 feet (15 m) wide were more susceptible to being reduced over time by human disturbance. Nearly all of the buffers they studied that were less than 50 feet (15 m) in width were significantly reduced in the few years the buffers had been present on the back of private lots. Some of the buffers were found to have been eliminated through complete clearing of native vegetation. Of the buffers wider than 50 feet (15 m), most still had some portion intact and overall they showed fewer signs of human disturbance.

In a study in the Monterey Bay area of California, Dyste (1995) examined 15 wetlands with buffers. All of the buffers suffered from human alteration including cutting of vegetation, soil compaction, and dumping of garbage.

5.5.5.2 Loss of Trees to Blowdown

In the Pacific Northwest, forested buffers are often “created” as leave-strips around wetlands or along streams when the surrounding forest is cleared for land development. These forested strips are then exposed to winter windstorms, which are common, often resulting in substantial loss of large trees due to blowdown.

Pollock and Kennard (1998) concluded that trees in narrow forested buffers (less than 76 feet [23 m] wide) have a much higher probability of suffering significant mortality from windthrow and blowdown than trees in wider buffers. They conclude that buffers in the range of 76 to 115 feet (23 to 35 m), created when the surrounding forest is cut, are the minimum width that can be expected to withstand the effects of wind in the long term.

5.5.5.3 Reduced Capacity for Sediment/Nutrient Removal

Many of the studies described earlier assessed the effectiveness of buffers in removing sediments and nutrients for short durations (on the order of one to two years, if the time period was discernable in the methods sections of the literature). One study that assessed water quality improvement over longer periods found that effectiveness diminished as the outer margins of the buffers became saturated with sediment (Dillaha and Inamdar 1997). Their findings suggest that buffers have a limited carrying capacity for sediment removal

and that larger buffers and other methods may be required to ensure long-term control of sediment.

Similarly, Todd (2000) cites work by Dillaha in 1993 that found less than 10% of grass filter strips were effective after three to five years. The grass filter strips became channelized and surface flows were no longer passing through as sheet flow that would allow contact with vegetation to remove sediments and nutrients. Todd emphasizes that, for buffers to be effective, they have to be sustainable over time and this must be a factor when determining buffer widths.

5.5.5.4 Summary of Key Points

- Human actions can reduce the effectiveness of buffers in the long term through removal of buffer vegetation, soil compaction, sediment loading, and dumping of garbage.
- Buffers may lose their effectiveness to disperse surface flows over time as flows create rills and channels, causing erosion within the buffer.
- Leaving narrow strips of trees can result in tree loss due to blowdown.
- Buffers may become saturated with sediment over time and become less effective at removing pollutants. The literature indicates that this should be considered when determining buffer widths.

5.5.6 Summary of Buffer Ranges and Characteristics from the Literature

The following discussion summarizes the many suggestions and recommendations in the literature for how buffer widths can be established. Many of these are found in synthesis documents that summarize scientific literature on buffers and then draw general conclusions. The recommendations in most of these syntheses take a more general approach and do not suggest how to determine buffer widths specifically in relation to desired functions such as protection of water quality or maintenance of habitat.

Several references (Castelle et al. 1992, Desbonnet 1994, Norman 1996, McMillan 2000, Todd 2000) identify four criteria that should be considered in determining the width of a buffer:

- The value of the aquatic resource to be protected by the buffer
- The characteristics of the aquatic resource in question, of the watershed contributing to the aquatic resource, and of the buffer itself
- The intensity of the adjacent land use (or proposed land use)

- The specific functions that the buffer is supposed to provide including the targeted species to be managed for and an understanding of their life-history needs

The feasibility or possibility of incorporating those four considerations into determining buffer dimensions is dependent upon the jurisdiction in question. Those same authors acknowledge that the scientific basis for determining the size of a buffer is often superseded by political expediency. Buffers are more often determined administratively as standard or fixed dimensions that may, or may not, be correlated with the criteria listed above.

Table 5-7 presents a summary of the buffer ranges recommended by the authors who conducted literature reviews or syntheses on buffer effectiveness. Minimums ranged from 25 feet (8 m) to 197 feet (60 m). Maximums ranged from 98 feet (30 m) for some land uses to 350 feet (107 m).

Table 5-7. Summary of recommendations for buffer dimensions from the literature.

Author(s)	Date	Minimum Buffer	Maximum Buffer	Comments
Castelle et al.	1994	25 feet (8 m)	98 feet (30 m)	"Adequate under most circumstances"
Desbonnet et al.	1994	49 feet (15 m)	98 feet (30 m)	Low-intensity land uses (agriculture, recreation, and low density residential)
		98 (30 m)	164 feet (50 m)	High-density residential housing and commercial/industrial
Fischer	2000	98 feet (30 m)	328 feet (100 m)	Larger buffer for reptiles, amphibians, birds and mammals
Groffman et al.	1991a	197 feet (60 m)	328 feet (100 m)	For most wildlife needs
Howard and Allen	1989	197 feet (60 m)		For most wildlife needs
McMillan	2000	25 feet (8 m)	350 feet (107 m)	Case by case, using a rating system and the intensity of proposed or existing land use
Norman	1996	164 feet (50 m)		To protect wetland functions; more may be required to protect more "sensitive wildlife species"

Table 5-8 outlines the general effectiveness of different buffer widths at removing pollutants and providing habitat (Desbonnet et al. 1994).

Table 5-8. A summary of pollutant removal effectiveness and wildlife habitat value of vegetated buffers according to buffer width (Desbonnet et al. 1994).

Buffer Width in Feet (m)	Pollutant Removal Effectiveness	Wildlife Habitat Value
16 feet (5 m)	Approximately 50% or greater sediment and pollutant removal	Poor habitat value; useful for temporary activities of wildlife
32 feet (10 m)	Approximately 60% or greater sediment and pollutant removal	Minimally protects stream habitat; poor habitat value; useful for temporary activities of wildlife
49 feet (15 m)	Greater than 60% sediment and pollutant removal	Minimal general wildlife and avian habitat value
66 feet (20 m)	Greater than 70% sediment and pollutant removal	Minimal wildlife habitat value; some value as avian habitat
98 feet (30 m)	Approximately 70% or greater sediment and pollutant removal	May have use as a wildlife travel corridor as well as general avian habitat
164 feet (50 m)	Approximately 75% or greater sediment and pollutant removal	Minimal general wildlife habitat value
246 feet (75 m)	Approximately 80% or greater sediment and pollutant removal	Fair to good general wildlife and avian habitat value
328 feet (100 m)	Approximately 80% or greater sediment and pollutant removal	Good general wildlife habitat value; may protect significant wildlife habitat
656 feet (200 m)	Approximately 90% or greater sediment and pollutant removal	Excellent general wildlife value; likely to support a diverse community
1,968 feet (600 m)	Approximately 99% or greater sediment and pollutant removal	Excellent general wildlife value; supports a diverse community; protection of significant species

Castelle et al. (1994), summarizing research conducted primarily before 1990, concluded "buffers necessary to protect wetlands and streams should be a minimum of 49 to 98 feet (15 to 30 m) in width under most circumstances." They note that the lower end of the spectrum is the minimum necessary to maintain physical and chemical processes, while the upper end of the spectrum may be necessary to maintain biological processes. This work appears to contradict the findings of Desbonnet et al. (1994). However, the language in the Castelle et al. report of 1994 states that the buffers should be a *minimum* of 49 to 98 feet; the report does not identify appropriate maximums. The report is most often quoted to imply that the full range of buffers should be *between* 49 and 98 feet.

McMillan (2000) recommends an approach to determining buffers that attempts to balance predictability with flexibility by setting standard buffer widths that can be altered on a case-by-case basis to adapt to site-specific factors. This approach for determining buffer width incorporates a rating system for wetlands, plus an assessment of the intensity of proposed or existing adjacent land use, to establish buffer widths ranging from 25 to 350 feet (8 to 107 m). It is perhaps the method that is closest to fitting the four bulleted

criteria outlined at the beginning of this section. It incorporates an understanding of the condition of the wetland, the buffer, and the proposed adjacent land use.

Several other authors also suggest that considering site-specific factors enhances the effectiveness of buffer strips over using fixed-width buffers (Haupt 1959, Steinblums et al. 1984). Belt and O'Laughlin (1994) note that, "The fixed minimum-width approach enjoys the virtue of simplicity in application, but has the potential for providing either not enough or too much protection."

Liquori (2000) also cautions against using fixed buffer widths to protect long-term ecological functioning of buffers and their associated aquatic resources. He notes that many of the functions that buffers provide are directly related to physical characteristics and biological processes within the buffers, and therefore, informed by site-specific information, a case-by-case argument could be made for establishing buffer widths. "The nature of the [functions a buffer provides] may significantly depend upon riparian structure both locally and as a mosaic over the watershed scale."

In urban settings, larger buffer widths are often prescribed in anticipation of future impacts from adjacent land use and activity upstream in the watershed. The most important criterion for determining buffer width is identification of the various functions the buffer is expected to provide. (Todd 2000.)

In agricultural lands, Welsch (1991) identifies a three-zone approach for establishing buffers:

- **Zone 1** consists of riparian-type trees and shrubs immediately adjacent to the stream, water body, or wetland. It should be a minimum 13 feet (4 m) wide, or adjusted to include the entire riparian area (the area with year-long or seasonal soil-moisture regime influenced by the stream or water body). Minimum length should be the length of the proposed disturbance outside the riparian management zones, or "the longest distance possible."

- **Zone 2** extends upslope from Zone 1 and consists of vegetation that may be periodically harvested as it matures. A minimum distance of 20 feet (6 m) should be allowed for this zone for small streams or water bodies; for larger streams or water bodies the total of Zones 1 and 2 can be increased up to 98 feet (30 m) or 30% of the geomorphic floodplain (whichever is less). Minimum length should match that of Zone 1. Zone 2 can be an active harvest zone, but trees and vegetation need to be left to provide soil holding and filtering capacity.

- **Zone 3** is added upslope of Zone 2 if adjacent land (away from the aquatic resource) is cultivated cropland or another land use with the potential for erosion or sediment production. Zone 3 is a vegetated filter strip and should be wide enough to control "concentrated flow erosion from cultivated cropland." Zone 3 vegetation should be established prior to the establishment of Zones 1 and 2.

This zonal approach is in response to proposed land use of active agriculture which implies regular creation of conditions with high erosion potential (grazing or tilling). It

also allows more active use of the central portion of the buffer and active management of the outer area of the buffer.

Townsend and Robinson (2001) build on this zonal approach and recommend guidance on maintenance of canopy coverage and closure. They suggest using species that readily resprout from stumps or roots in the areas nearest the stream channels (to allow the vegetation to respond to flood damage and/or beaver activity). They stress the need for ongoing maintenance, especially in Zone 3, to ensure that erosive flows are not causing rills or channelized flows into Zone 2. They also note that while most of these buffers will be applied on an ownership basis, greater benefit would be realized if the concept of zoned buffers were applied on a watershed basis.

Other recommendations are based on wildlife species of particular interest. Based on their study of waterbirds in Florida, Rodgers and Smith (1997) recommend a buffer width of 328 feet (100 m) to ensure that birds will not be triggered into an "approach" response, a state which occurs prior to actual flushing. They derived this figure by analyzing the flushing distance from human approach for 16 species, then adding 131 feet (40 m) to that distance. The 131-foot (40 m) distance was derived from previous work which found that birds became alert (stopped their ongoing behavior and focused on the approaching human) in a range of 82 to 131 feet (25 to 40 m).

5.5.6.1 Summary of Key Points

- Several researchers have recommended using four basic criteria to determine the width of a buffer:
  - the value of the aquatic resource to be protected by the buffer;
  - the characteristics of the aquatic resource in question, of the watershed contributing to the aquatic resource, and of the buffer itself;
  - the intensity of the adjacent land use (or proposed land use); and
  - the specific functions that the buffer is supposed to provide including the targeted species to be managed for and an understanding of their life-history needs.
- Protecting wildlife habitat functions of wetlands generally requires larger buffers than protecting water quality functions of wetlands.
- Effective buffer widths should be based on the above factors and generally should range from: 25 to 75 feet (8 to 23 m) for wetlands with minimal habitat functions and adjacent low-intensity land uses; 50 to 150 feet (15 to 46 m) for wetlands with moderate habitat functions or adjacent high-intensity land uses; and 150 to 300 feet (46 to 92 m) for wetlands with high habitat functions.
- Fixed-width buffers may not adequately address the issues of habitat fragmentation and population dynamics. Several researchers have recommended

a more flexible approach that allows buffer widths to be varied depending on site-specific conditions.

5.6 Chapter Summary and Conclusions

Wetlands are defined using well established language that is generally consistent between federal and state laws. However, certain wetland types are sometimes excluded from regulation. These include small wetlands, isolated wetlands, and Prior Converted wetlands. The scientific literature makes clear that small wetlands and isolated wetlands provide important functions and does not provide any rationale for excluding these wetlands from regulation. Little scientific information is available on Prior Converted wetlands, but there is no evidence to suggest that they are unimportant in providing wetland functions.

Wetland delineation is conducted according to either the federal or state delineation manual. These manuals are consistent and, when applied correctly, will result in the same wetland boundary. Wetland rating systems are a useful tool for grouping wetlands based on their needs for protection. The most widely used method in Washington is the state's rating system which places wetlands in categories based on their rarity, sensitivity, irreplaceability, and functions.

Wetland buffers are a critical tool for protecting wetland functions. Findings regarding buffer functions and effectiveness are related to what functions are in question, what land use activities are being buffered, and the characteristics of the wetland and the buffer itself.

The literature confirms that for water quality improvement (e.g., sediment removal and nutrient uptake) there is a non-linear relationship between buffer width and increased effectiveness. Sediment removal and nutrient uptake are provided at the greatest rates within the immediate outer portions of a buffer (nearest the source of sediment/nutrient), with increasingly larger widths of buffers required to obtain measurable increases in those functions. Additionally, the long-term effectiveness of buffers in providing such mechanical and biological processes is not well documented in the literature and may represent a critical need for future research.

For buffer functions related to protection of wildlife species and populations that use wetlands, the literature has documented the need for significantly larger buffers than those adequate to provide sediment removal and nutrient uptake. Research confirms that many wildlife species and guilds are dependent upon wetlands for only portions of their life cycles and they require upland habitats adjacent to the wetland to meet all their life needs. Some species use upland habitats that are far from the source wetland. The literature documents that, without access to appropriate upland habitat and the opportunity to move between habitats across a landscape, it is not possible to maintain viable populations of many species. Beyond simply providing adequate upland habitat adjacent to a single wetland, the literature on the maintenance of wildlife populations

finds that it is necessary to link habitat types, including wetlands and uplands, across a landscape in order to maintain genetically viable populations.

Several authors who suggested recommendations for buffer widths based on their own synthesis of the literature have recommended variable widths based on the conditions of the wetland, the conditions of the buffer, the proposed land uses adjacent to the buffer, and what functions are intended to be managed. For protection and maintenance of wildlife habitat functions of wetlands, these studies suggest that effective buffer widths should be based on the above factors and generally should range from: 25 to 75 feet (8 to 23 m) for wetlands with minimal habitat functions and adjacent to low-intensity land uses; 50 to 150 feet (15 to 46 m) for wetlands with moderate habitat functions or high-intensity land use that is adjacent; and 150 to 300 feet (46 to 92 m) for wetlands with high habitat functions.

Chapter 6 continues the discussion of regulatory tools used to manage wetlands by discussing wetland compensatory mitigation and its effectiveness.

Chapter 6
The Science and Effectiveness of Wetland Mitigation

6.1 Reader's Guide to this Chapter

This chapter synthesizes the scientific literature regarding compensatory mitigation and its effectiveness at reducing the severity of activities that detrimentally affect wetlands. It also reports the suggestions made by various authors regarding ways to improve compensatory mitigation.

6.1.1 Chapter Contents

Major sections of this chapter and the topics they cover include:

Section 6.2, Introduction and Background to Wetland Mitigation describes the process of wetland mitigation, which encompasses a series of steps that are performed sequentially. Compensation for wetland impacts is just one of these steps.

Section 6.3, Success of Compensatory Mitigation Wetlands synthesizes the literature on the biological, ecosystem, or functional success of compensatory mitigation projects. This section does not specifically evaluate the successful compensation for wetland area; that is discussed in Section 6.7.

Section 6.4, Compliance with Permit Requirements describes studies that evaluated several aspects of how well compensatory mitigation projects met legal or permit requirements. These included whether projects were completed or installed according to plan, whether they attained the required wetland acreage, whether performance standards were achieved, whether the project was monitored or maintained, and whether the regulatory agencies tracked the project.

Section 6.5, Types of Compensatory Mitigation discusses the use and effectiveness of restoration, creation, enhancement/exchange, preservation, mixed compensatory mitigation, mitigation banking, and in-lieu fees.

Section 6.6, Replacement Ratios describes the rationale for the use of ratios in determining the acreage required as compensation for a given area of wetland impact. It synthesizes the literature on the ratios that were required and those actually achieved for numerous projects and it discusses approaches proposed in the literature to more effectively determine compensatory mitigation ratios.

Section 6.7, Replacement of Wetland Acreage summarizes the results of studies examining whether compensatory wetland mitigation is actually replacing the acreage of wetland losses authorized. This includes both evaluations of overall permitting programs and of specific compensation projects in compensating for wetland acreage.

Section 6.8, Functions and Characteristics Provided by Created, Restored, or Enhanced Wetlands describes the ability of mitigation wetlands to provide for wildlife habitat, plant communities, adequate soil conditions, and water quality/quantity functions. Compensation wetlands are often compared with pre-existing or reference wetlands in these studies.

Section 6.9, Reproducibility of Particular Wetland Types summarizes the literature regarding whether and how easily certain wetland types, such as bogs, fens, vernal pools, alkali wetlands, and mature forested wetlands, can be reproduced or restored.

Section 6.10, Suggestions from the Literature for Improving Compensatory Mitigation summarizes numerous recommendations made by researchers to improve the success of compensation projects—ranging from improvements to regulations and site selection, to better performance standards, to a broader landscape approach, to mitigation banking.

Section 6.11, Chapter Summary and Conclusions ties together the major concepts presented in the chapter.

6.1.2 Where to Find Summary Information and Conclusions

Each major section of this chapter concludes with a brief summary of the key points resulting from the literature review on that topic in a bullet list format. The reader is encouraged to remember that a review of the entire section preceding the summary is necessary for an in-depth understanding of the topic.

For summaries of the information presented in this chapter, see the following sections:

- Section 6.3.2
- Section 6.4.9
- Section 6.5.8
- Section 6.6.4
- Section 6.7.3
- Section 6.8.6
- Section 6.9.5
- Section 6.10.7

In addition, Section 6.11 provides a summary of the chapter and conclusions about the overarching themes gleaned from the literature and presented in this chapter.

6.1.3 Data Sources and Data Gaps

The synthesis in this chapter is based on more than 50 articles, government reports, and conference proceedings that have been published since about 1990 on the topic of compensatory mitigation. (The literature did not address the other types of mitigation listed in Section 6.2.1.)

The articles and reports reviewed used a variety of terms to define what they were assessing or evaluating. For the purposes of this synthesis, "effectiveness" is used as a general term referring to how compensatory wetland mitigation is doing overall, including evaluations of success, compliance, and functions and characteristics. These terms will be defined more precisely in subsequent sections.

Data from a variety of sources are summarized throughout this chapter in a series of tables. To simplify the tables and maximize space, each literature source listed in the tables is represented by a reference number listed in Table 6-1. This is not a comprehensive list of all references cited in this chapter; see the references section at the end of Volume 1 for a complete list of literature sources.

Location of studies cited in this chapter

The articles and reports that evaluated the effectiveness of individual compensatory mitigation projects focused on a variety of locations, including Washington, Oregon, California, Louisiana, Michigan, Indiana, Ohio, New Jersey, Massachusetts, Maryland, Tennessee, and Florida.

Studies that assessed specific wetland functions of mitigation and non-regulatory restoration sites were located in: Washington, Oregon, Wyoming, Iowa, Minnesota, Wisconsin, Illinois, Ohio, New York, Pennsylvania, Connecticut, West Virginia, South Carolina, Florida, Canada, Sweden, Spain, Austria, and central Europe.

Table 6-1. Literature sources and corresponding reference numbers.

Reference No.	Literature Source	Reference No.	Literature Source
1	Allen and Feddema (1996)	19	Shaich and Franklin (1995)
2	Balzano et al. (2002)	20	Storm and Stellini (1994)
3	Brown and Veneman (2001)	21	Torok et al. (1996)
4	De Weese (1998)	22	Wilson and Mitsch (1996)
5	Erwin (1991)	23	Barry et al. (1996)
6	Gwin and Kentula (1990)	24	Castelle et al. (1992)
7	Holland and Kentula (1992)	25	Celedonia (2002)
8	Holland and Bossert (1994)	26	Chovanec (1994)
9	Johnson et al. (2000)	27	Hunt et al. (1999)
10	Johnson et al. (2002)	28	Kentula (2000)
11	Jones and Boyd (2000)	29	National Research Council (2001)
12	Kentula et al. (1992)	30	Race and Fonseca (1996)
13	Kunz et al. (1988)	31	Sheldon and Dole (1992)
14	McKinstry and Anderson (1994)	32	Whittecar and Daniels (1999)
15	Michigan Department of Environmental Quality (2000)	33	Zedler and Callaway (2000)
16	Mockler et al. (1998)	34	Mitsch and Wilson (1996)
17	Morgan and Roberts (1999)	35	Breaux and Serefiddin (1999)
18	Robb (2002)	36	Sudol and Ambrose (2002)
		37	Cole and Shafer (2002)

6.2 Introduction and Background to Wetland Mitigation

6.2.1 Wetland Mitigation Sequence

Mitigation is a sequential process used to reduce the severity of effects from activities that potentially affect wetlands. When a land use project has the potential to adversely affect a wetland, the federal, state, and/or local government agency regulating the wetland will initiate the process of mitigation.

According to the rules implementing the Washington State Environmental Policy Act (Chapter 197.11 WAC), mitigation involves the following steps that are performed sequentially:

1. *Avoiding the impact altogether by not taking a certain action or parts of an action;*

2. *Minimizing impacts by limiting the degree or magnitude of the action and its implementation, by using appropriate technology, or by taking affirmative steps to avoid or reduce impacts;*

3. *Rectifying the impact by repairing, rehabilitating, or restoring the affected environment;*

4. *Reducing or eliminating the impact over time by preservation and maintenance operations during the life of the action;*

5. *Compensating for the impact by replacing, enhancing, or providing substitute resources or environments; and/or*

6. *Monitoring the impact and taking appropriate corrective measures* (WAC 197.11.768).

Avoidance is a mitigation measure that eliminates the direct loss of wetland area and functions. Avoidance does not, however, eliminate indirect losses of wetland function. For example, consider a hypothetical proposal to develop a 5-acre parcel of land. The parcel contains 2 acres of wetland. The development is designed around the wetland and will therefore avoid any direct loss. Avoidance as a mitigation measure has been satisfied. Yet if buildings and parking lots surround the wetland, indirect impacts to wildlife habitat and likely hydrology will have occurred in the form of isolation, fragmentation, and altered hydroperiod.

Minimization, as a mitigation measure, reduces the direct loss of wetland acreage and functions by redesigning or scaling back a development to lessen the amount of wetland acreage altered. Minimization may still result in the indirect loss of functions.

Projects that will result in the temporary alteration of a wetland, such as during installation or maintenance of an underground pipeline, typically utilize rectification as a mitigation measure. In the case of an underground pipeline that crosses through a wetland, vegetation, soil, and water movement would probably be disturbed and altered. Rectification would entail replacing the soil, restoring the water movement, and replanting or seeding the vegetation. The wetland acreage and functions are temporarily lost during construction.

Mitigation measures to reduce and monitor the impacts to wetlands may be beneficial in limiting indirect effects from development. The scientific literature reviewed for this synthesis document did not contain information on the use or effectiveness of any of the

mitigation measures defined above, except compensatory mitigation, which is the focus of the remainder of this chapter.

6.2.2 The Emergence of Compensatory Wetland Mitigation

The term "compensatory mitigation" refers to the compensation stage of the sequential mitigation process (number 5 in the list of steps described earlier). For wetlands, it typically involves producing new wetland area, functions, or both as compensation for wetland area, function, or both that have been or will be lost due to a permitted activity. Compensatory wetland mitigation generally entails performing one or more of the following types of compensation:

> The term **compensatory mitigation** refers to the compensation stage of the mitigation process (number 5 in the list of steps above). Because the regulatory requirements and policies tend to focus on the compensation stage, the term "mitigation" is often used to refer to compensation, which is just one part of the overall mitigation process.

- Restoring wetland conditions (and functions) to an area
- Creating new wetland area and functions
- Enhancing functions at an existing wetland
- Preserving an existing high-quality wetland to protect it from future development

The use of compensatory mitigation for wetland loss emerged in the 1980s (Roberts 1993, National Research Council 2001). The U.S. Army Corps of Engineers considered the process of mitigation as part of the National Environmental Policy Act of 1969. However, it wasn't until 1980 when the Environmental Protection Agency (EPA) issued new guidelines for Section 404(b)(1) of the Clean Water Act that mitigating for wetland losses by creating or restoring another wetland as compensation became widely acceptable (National Research Council 2001). Compensatory mitigation was seen as a way to speed up an arduous process of documenting avoidance and minimization efforts, while satisfying concerns about the loss of ecosystems and functions (Roberts 1993). Creating or restoring wetland area to compensate for permitted wetland losses was viewed and publicized as a way to allow development while preventing a net loss of wetland areas.

By the late 1980s, studies of the effectiveness of compensatory mitigation were emerging, with mixed results. The primary indication was that replacing or replicating an existing wetland was difficult, if not impossible (Kusler and Kentula 1990, National Research Council 2001). However, some wetland types and functions could be approximated given the proper conditions (Kusler and Kentula 1990, National Research Council 2001). This chapter focuses on studies published since 1990 that examined the effectiveness of compensatory mitigation.

6.3 Success of Compensatory Mitigation Wetlands

Compensatory mitigation "success" is poorly defined and often contentious (Kentula 2000). The literature refers to legal success, biological success, ecosystem success (Wilson and Mitsch 1996), functional success (Mockler et al. 1998), or some combination of these.

Compliance generally means the same as **legal success**. It is evaluated by comparing the actual on-the-ground, or as-built, conditions against what was required in the permit. Studies describing legal success are referred to as *compliance* in this document, and they are discussed in Section 6.4.

Biological, ecosystem, or functional success is the focus of this section and, therefore, when the term *success* is used herein, it refers exclusively to biological/functional success. Success involves an evaluation of the factors that characterize a wetland (e.g., hydroperiod, vegetation, soils), the performance of functions, or both. Best professional judgment, one of a variety of function assessment methods, or both are used by researchers to evaluate this type of success.

A compensation site may comply with all its permit requirements and not replace the functions or values of the wetland that were lost. On the other hand, a site may fall short of meeting its permit requirements and yet still be a functioning wetland. For example, a site may not attain all its goals or performance standards, yet it may still provide a variety of important wetland functions. Furthermore, a mitigation site may not be in compliance or achieve biological success at the particular time of an evaluation, but later it may meet all relevant criteria.

Rather than judging the success or failure of a compensatory wetland mitigation project at a single point in time, Zedler and Callaway (2000) proposed evaluating how a project progresses over time. The authors suggest that a focus on progress would encourage proponents to acknowledge problems occurring at a site and look for solutions. Zedler (2000) proposes that more compensation projects should be viewed as experiments without a specific desired outcome. In lieu of attaining a specific level of performance, projects would be monitored as experiments for at least 25 years. The regulatory framework currently in place, however, does not support this method of evaluation due to the relatively short timeframe allowed for monitoring and assessing the compliance of compensation projects (Breaux and Serefiddin 1999, Zedler 2000).

6.3.1 Results of Literature Studies

Several studies determined the level of success of compensatory mitigation projects (Table 6-2). Though the data indicated that some projects were successful and some projects were unsuccessful, most compensation projects had an intermediate level of success; they were neither fully successful nor completely unsuccessful.

- 25 to 66% of projects were determined to have an intermediate level of success

- 13 to 43% of projects achieved full success
- 7 to 97% of projects were unsuccessful, though half of the studies found that at least 20% of projects were unsuccessful

The methods used to evaluate the success of compensatory wetland mitigation projects varied from best professional judgment (Storm and Stellini 1994) to function assessments (Wilson and Mitsch 1996, Balzano et al. 2002), to quantitative measures of vegetation cover and survival (Allen and Feddema 1996), or some combination (Michigan Department of Environmental Quality 2000, Johnson et al. 2002). Though the methods of evaluation differed, most studies considered similar variables such as wetland area, hydrologic conditions, wildlife suitability, vegetation, and soils.

Table 6-2. Results of studies examining the success of compensatory mitigation.

Location of Study and Reference No. [a]	No. of Projects Evaluated	Level of Success	Evaluation Criteria
Washington State (10)	24	13% fully successful 33% moderately successful 33% minimally successful 21% not successful	Wetland acreage, performance standards, goals/objectives, contribution to functions, comparison with wetland lost
Washington/ King County (16)	38	3% successful 97% not successful	Replacing functions
Western Washington (20)	17	23% functioned well ecologically 65% functioned poorly 12% were not completed	Vegetation diversity, non-native plant dominance, structural diversity, wildlife use, adjacent land uses, vegetation cover vs. open water
Southern California (1)	75	32 successful 9 mostly successful 10 half successful 5 unsuccessful 8 under construction 5 not initiated 6 did not require mitigation	Project installed according to plan; percent cover of vegetation (dead, living, and invasive)
California/ Orange County (36)	55	16% successful 58% partially successful 26% failures	Qualitative evaluation based on habitat quality (e.g., veg. density and diversity, invasive species, tree height)

Location of Study and Reference No. [a]	No. of Projects Evaluated	Level of Success	Evaluation Criteria
Ohio (22)	5	1 high 2 medium to high 1 medium 1 medium to low	WETII evaluation (Adamus et al. 1989) - hydrology, soils, vegetation, wildlife, water quality
New Jersey (2)	74	Wetland Mitigation Quality Assessment scores were indexed from 0 (low) to 1 (high). The average score was 0.51, and the range was 0.25 to 0.83	Hydrology, soils, vegetation, wildlife suitability, site characteristics, and landscape features
Michigan (15)	69	22% successful overall 78% unsuccessful overall	Project's legal rating (permit compliance) and biological rating (wetland acreage). Does not include enhancement

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

6.3.2 Summary of Key Points

- Success can be defined as meeting biological or ecological criteria, which may include an assessment of functions (legal success or regulatory compliance will be discussed in Section 6.4).
- The majority of compensatory wetland mitigation projects were found to be neither fully successful nor completely unsuccessful, but somewhere in between, relative to biological or ecological functions.
- Though the methods used to evaluate project success differed, the studies considered similar parameters, such as vegetation, soils, and hydrology.

6.4 Compliance with Permit Requirements

Regulatory agencies typically require wetland compensation for authorized, unavoidable, wetland impacts. A wetland mitigation plan is reviewed and approved as part of the permit approval process. The wetland mitigation plan outlines how wetland impacts will be compensated for. The mitigation plan should identify how the project will be designed, as well as addressing wetland acreage, hydroperiod, vegetation, goals, objectives, performance standards, monitoring, maintenance, and contingency actions. These are the parameters by which regulators measure compliance.

According to the Merriam-Webster dictionary, compliance means "conformity in fulfilling official requirements." Regarding compensatory wetland mitigation, compliance means that a project has satisfied or is satisfying the legal requirements and obligations identified in a permit.

Most studies that examined compliance investigated how well a compensatory wetland mitigation project complied overall (i.e., with all applicable permit requirements). Several of these studies only reported the results of the overall evaluations. Other studies evaluated how well projects complied with individual requirements, such as:

- Installation – whether the project was installed
- Installation according to plan – whether the project was constructed according to the approved mitigation plan and design
- Wetland area establishment – whether the project obtained the acreage of wetland that was required
- Performance standard/goals/objectives attainment – whether the project performed as anticipated
- Monitoring – whether the project was monitored as required (or was required to be monitored)
- Maintenance – whether project maintenance was performed (or required)
- Regulatory follow-up – whether any regulatory agencies made an attempt to track the project after the permit was issued

Each of these types of evaluations is discussed in subsequent sections below.

6.4.1 Compliance Overall

Several studies attempted to determine how well a project complied with several or all of its permit requirements. Because permit requirements vary by state and over time, not all compliance evaluations considered the same criteria or requirements. Where specified, the requirements evaluated by a given study are identified in Table 6-3.

Twelve studies evaluated overall compliance with regulatory requirements for compensatory wetland mitigation projects (Table 6-3). In Washington State four studies evaluating compliance have been conducted in the past decade (Storm and Stellini 1994, Mockler et al. 1998, Johnson et al. 2000, Johnson et al. 2002), and two studies have been conducted in Oregon (Gwin and Kentula 1990, Shaich and Franklin 1995).

The studies in Washington found that less than one-third of compensation projects complied with their regulatory requirements. In Oregon studies revealed that compliance of projects ranged from zero to 36%.

Studies from other states demonstrated more variability in levels of compliance. Results ranged from less than 20% to about 80% of projects in compliance (Holland and Bossert 1994, De Weese 1998, Morgan and Roberts 1999, Michigan Department of Environmental Quality 2000, Brown and Veneman 2001, Balzano et al. 2002, Sudol and Ambrose 2002).

More recent studies (published in 2000 or after) did not report higher levels of compliance than studies conducted in the 1990s. One might therefore assume that compensation projects have not improved over the years. However, it is important to realize that as knowledge of wetland science and compensatory mitigation has improved and evolved, permit requirements have likewise evolved (Kentula 2000). More recent studies may be evaluating compensation projects that are being held to a higher standard than projects permitted and evaluated in the 1990s (Sudol and Ambrose 2002). However, a study by Cole and Shafer (2002) in Pennsylvania observed that permit requirements had not changed noticeably over the 14-year range of permits they evaluated (1986-1999).

Table 6-3. Level of overall compliance of compensation projects.

Location of Study and Reference No. [a]	No. of Projects Evaluated	% of Projects in Compliance with all Requirements	Evaluation Criteria
Washington (9)	45	29%	• Project installed • Installed according to plan • Meet performance standards
Washington (10)	24	29%	• Establish required wetland acreage • Meet performance standards • Meet goals/objectives
Washington/western (20) [b]	17	18%	• Installation of both development and compensatory mitigation projects as required
Washington/King County (16) [c]	29 (38)	21% (16%)	• Meet performance standards (project installed)
Oregon/Portland metro area (19) [d]	72	36%	• Project installed • Upland buffer area/vegetation requirements • Requirements for timing of project construction • Wetland vegetation requirements • Hydrology requirements • Requirements for water control structures • Fencing requirements
Oregon/Portland metro area (6)	11	0%	• Construction plans match permit specs • As-built matches permit specs: wetland area/shape • Actual slopes match planned slopes • Vegetation established as planned
California/ Orange County (36) [e]	57	53%	• Project installed • Meet performance standards/ permit conditions
California/ vernal pools (4)	25	83%	• Attaining performance standards required by Corps

Location of Study and Reference No. [a]	No. of Projects Evaluated	% of Projects in Compliance with all Requirements	Evaluation Criteria
Massachusetts (3) [f]	109 (7)	43% (100%)	• Project installed • Compensation project of required size • Water inputs sufficient for wetland conditions • At least 75% cover wetland plants (FAC or wetter)
Tennessee (17)	50	12%	• Establish required acreage of wetland • Meet performance standards
Michigan (15) [g]	74	18%	• Mitigation acreage requirement • Implementation of approved mitigation plan • Conservation easement • Submittal of as-built plans • Monitoring • Placement of elevated wildlife structures • Construction schedule with specified completion date • Prohibited actions • Corrective measures identified • Financial assurances
Louisiana (8)	9	78%	• Meet Corps of Engineers permit conditions
New Jersey (2) [h]	88	48% weighted average	• Grading (56% concurrence) • Hydrology (47% concurrence) • Soil (51% concurrence) • Vegetation cover (39% concurrence) • Vegetation survival (28% concurrence) • Design (56% concurrence)

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

[b] Compliance not determined for 53% of projects due to lack of information.

[c] 38 projects examined; 9 not completed. Compliance information for 38 projects is in parentheses.

[d] Not all projects had requirements for all criteria (e.g., only 8% had requirement for fencing).

[e] Calculated from data provided.

[f] 5 projects did not result in wetland impact and were subtracted from the project total. Results were recalculated from the data provided. Parentheses = data for variance projects (received more oversight).

[g] Permit conditions from criteria list were considered if specified in permit.

[h] Evaluated concurrence with applicable criteria. Percent = average concurrence score for 88 projects. Average concurrence score for each criterion provided in parentheses.

6.4.2 Project Installation

A number of studies inventoried or randomly selected mitigation projects from a permitting database to determine whether they had been constructed or installed at all. Four studies were conducted in Washington. Seven other states, including Oregon, also investigated whether mitigation projects had been installed.

Results indicated that most projects were installed (Table 6-4). The studies from Washington found that 74 to 93% of compensatory mitigation projects had been installed. Studies from most of the other states show similar results (64 to 99%). However, studies performed in Florida and Tennessee revealed that less than half of the compensatory wetland mitigation projects had been installed (Erwin 1991, Morgan and Roberts 1999).

The relatively high percentage of projects that were installed implies that the low levels of overall compliance result from inadequate design, installation, maintenance, follow-up, or some combination.

Table 6-4. Percent of compensatory mitigation projects that were installed.

Location of Study and Reference No. [a]	No. of Projects Evaluated	Percent of Projects Installed
Washington (9)	45	93%
Washington/ King County (16)	38	76%
Washington/ western (20)	17	88%[b]
Washington (13)	35	74%
Oregon/ Portland metro area (19)	90	99%
California/ southern (1)	75	93%
California/ Orange County (36)	57	96%[b]
Michigan (15)	159	85%
Indiana (18)	333	64%
Massachusetts (3)[c]	109	77%
Tennessee (17)	100	47%
Florida (5)	NA	~40% [d]

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

[b] Calculated from data provided.

[c] Five projects did not result in wetland impact and were subtracted from the project total. Results were recalculated from the data provided.

[d] "Out of more than 100 permitted projects requiring wetland mitigation only 40 had undertaken any mitigation activity."

6.4.3 Installation According to Plan

Another aspect of determining mitigation compliance is evaluating whether a mitigation project has been installed according to its approved plan. When compensatory wetland mitigation is necessary to offset proposed wetland losses, regulatory staff generally require a wetland mitigation plan/report. The mitigation plan should provide specific information about project construction, including detailed design drawings. Approval of a permit for wetland loss is often contingent upon approval or acceptance of the wetland mitigation plan/report.

It is commonly assumed that a project will be built exactly as it is designed. However, many factors during construction and installation can influence what is actually built on the ground. Therefore, permit requirements often require (or recommend) submittal of an as-built plan/report that documents the final installed conditions of a site after construction is complete. When available, as-built drawings are used to document the baseline conditions for monitoring of a site.

Three studies evaluated whether compensation projects were installed according to approved plans (Table 6-5). Results from both Washington and New Jersey indicate that more than half of the compensatory mitigation projects were installed according to requirements (Johnson et al. 2000, Balzano et al. 2002). Johnson et al. (2000) found that 88% of the projects that submitted an as-built plan/report were installed according to plan. A study in Oregon, however, determined that none of the projects were implemented according to plan (Gwin and Kentula 1990). All three studies mentioned grading and vegetation as the elements of the plan/design that were not implemented correctly.

The divergent results might be the result of time (i.e., when the project was designed and permitted). For example, the projects reviewed by Gwin and Kentula (1990) were designed, permitted, and constructed in the early 1980s. Since that time much has been learned by those who design, construct, and regulate compensatory mitigation projects. It is possible that improved designs, experience and skill in implementing the designs, and improved regulatory follow-up have resulted in a higher percentage of projects being installed according to plan by the mid- to late 1990s. However, the current scientific literature does not address this possibility.

Table 6-5. Percent of compensatory mitigation projects installed according to plan.

Location of Study and Reference No. [a]	No. of Projects Evaluated	Percent Installed to Plan	Aspects Not Installed to Plan
Washington (9)	42	55%	Mainly vegetation, also grading, miscellaneous plan elements (e.g., fences, signs)
Oregon/ Portland metro area (6)	11	0%	Size, shape, slopes, and vegetation
New Jersey (2)	88	56%	Incorrect elevations, sizes, and/or shapes

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

6.4.4 Establishment of Wetland Acreage

Compensatory wetland mitigation projects are intended to compensate for the loss of wetland area and functions. Hence, permits and mitigation plans often identify a specific acreage of compensation required to offset those losses. Establishing the required acreage is therefore a critical criterion of regulatory compliance. (Functions provided by compensatory mitigation projects are discussed later in this chapter.)

Thirteen studies examined compensatory wetland mitigation sites to determine if the acreage of wetlands required by the permits had been established (Table 6-6). The studies presented the data from these investigations in two ways:

- **The percentage of projects establishing the required wetland acreage.** Researchers determined if each project met its required wetland acreage, then reported how many projects actually met the wetland acreage requirement as a percentage of the total number of projects considered. A few studies mentioned a specific threshold, such that a project had to be significantly smaller than required in order to fail to meet its wetland area (Brown and Veneman 2001, Johnson et al. 2002, Morgan and Roberts 2003).

- **The percentage of compensatory wetland acreage established.** Researchers determined the total acreage of compensatory mitigation that was verified as wetland for all the projects considered. The study then reported the total acreage of wetland compensation that was established as a percentage of the total acreage that was required for all the projects considered.

Over half of projects achieved the required wetland area in Washington and Oregon (Shaich and Franklin 1995, Johnson et al. 2002). In fact, the majority of studies determined that about half of the compensation projects established the required acreage of wetland. However, three studies found that less than 30% of projects met their acreage requirements (McKinstry and Anderson 1994, Balzano et al. 2002, Morgan and Roberts 2003). In New Jersey only 7% of projects achieved the wetland acreage requirements (Balzano et al. 2002).

For the total acreage of wetland achieved versus required, a study from Washington determined that 84% of the required acreage of compensatory wetlands was established (Johnson et al. 2002), while a study in Oregon found about 70% of the required wetland acreage was established (Gwin and Kentula 1990). Results from other states indicated between 44 and 74% of the required wetland acreage had been established.

Why is there a discrepancy between the percent of projects achieving acreage and the percentage of total acreage established? New Jersey, for example, found that only 7% of compensation projects achieved the required wetland acreage, yet 63% of the total required wetland acreage was established. This is probably due to small, individual projects that establish a portion of the required acreage but fall short of the total amount required. For example, a site that was required to provide 1 acre of mitigation but only provided 0.8 acre would not meet the acreage criteria. However, the 0.2-acre difference may represent a very small fraction of the total acreage of compensation evaluated for a large study, thereby affecting the total acreage percentage very little.

Table 6-6. Establishment of required wetland acreage.

Location of Study and Reference No. [a]	No. of Projects Evaluated	% of Projects Achieving Required Wetland Area	% of Required Wetland Area that Was Established
Washington (10) [b]	24	58%	84%
Oregon/Portland metro area (19) [c]	72	53% [f]	NA
Oregon/Portland metro area (6)	11	NA	71%
California/southern (1) [d]	75	NA	69%
California/Orange County (36)	55	52%	NA
Wyoming (14)	64	14% [f]	NA
New Jersey (2)	85	7%	63%
Tennessee (17)	50	28%	68%
Ohio (22)	5	40%	66%
Indiana (18)	31	NA	44%
Michigan (15)	159	50%	NA
Massachusetts (3) [e]	109	46%	NA
Florida (5)	NA	NA	74%

NA= information not available

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

[b] West of Cascades, projects established 92% of required acreage; east side projects established 25% of required acreage.

[c] Compensation wetlands were 16 acres (6.5 ha) short of the 69 acres required.

[d] Projects > 8.5 acres (3.4 ha) resulted in a net gain of 17 acres (6.9 ha) of wetland area, while projects < 8.5 acres resulted in a net loss of almost 25 acres (10 ha).

[e] Five projects did not result in wetland impact and were subtracted from the project total. Results were recalculated from the data provided.

[f] Calculated from data provided.

6.4.5 Attainment of Goals, Objectives, and Performance Standards

Another critical component of compliance for a compensatory wetland mitigation project is determining whether the project has met its goals, objectives, and/or performance standards. Goals, objectives, and performance standards are generally included as part of an approved wetland mitigation plan. Goals and objectives are intended to provide a blueprint for what the project proposes to accomplish in terms of anticipated wetland type, specific habitat, functions, and/or values. The performance standards are intended to provide measurable criteria to determine if the project has accomplished its goals and objectives (Hruby et al. 1994, Ossinger 1999).

Two separate factors were investigated in the studies reviewed:

- Whether a project had goals, objectives, and performance standards
- Whether projects were meeting their goals, objectives, and performance standards

Data in Table 6-7 indicate that at least three-quarters of projects had goals, objectives, or both (Erwin 1991, Storm and Stellini 1994, Johnson et al. 2002). However, fewer projects met the goals/objectives (10 to 38%) according to the two studies that reported this information (Erwin 1991, Johnson et al. 2002).

In general, performance standards were specified less frequently than goals and objectives, though at least half of the projects had them (Erwin 1991, Storm and Stellini 1994, Mockler et al. 1998, Johnson et al. 2000, Cole and Shafer 2002). Two studies conducted in Washington determined that 21% of projects met their performance standards (Mockler et al. 1998, Johnson et al. 2002), while a third study from Washington found that 35% of projects met performance standards (Johnson et al. 2000).

The percent of projects that had performance standards appeared to increase with more recent projects. For example, Storm and Stellini (1994) and Cole and Shafer (2002) evaluated compensation projects that were permitted in the mid to late 1980s or early 1990s. Performance standards may not have been as rigorously required (Cole and Shafer 2002) or they may not have been specifically identified as performance standards. For example, of 10 projects that did not contain performance standards, 30% were permitted in the late 1980s and 80% were permitted prior to 1995, while 20% were permitted in the late 1990s (Cole and Shafer 2002).

Time does not appear to be a factor in whether projects met their performance standards. Cole and Shafer (2002) did not find that performance standards noticeably changed in terms of content from projects permitted in the late 1980s to the late 1990s. The more recent projects did not appear any more likely to meet performance standards than earlier projects.

More information on performance standards is provided in Section 6.10.4.

Table 6-7. Attainment of goals, objectives, and performance standards.

Location of Study and Reference No. [a]	% Projects w/ Goals or Objectives	% Projects w/ Performance Standards	% Projects Meeting Goals or Objectives	% Projects Meeting Performance Standards
Washington (9)	NA	87% [b]	NA	35%
Washington (10)	92%	NA	38%	21%
Washington/ King County (16)	NA	100%	NA	21%
Washington/ western (20)	76%	53%	NA	NA
Pennsylvania (37)	NA	57%	NA	62%
Florida (5)	85%	60%	10%	NA

NA = information not available.

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

[b] Calculated from data provided.

6.4.6 Monitoring

To determine if a compensatory wetland mitigation project is in compliance, it is necessary to monitor the project over time. Monitoring requirements are typically identified in the wetland mitigation plan. The duration, frequency, and methods of monitoring depend on the goals, objectives, and performance standards for the project.

Monitoring is the process through which data about site conditions are gathered. Monitoring data are used to determine whether a project is achieving its performance standards, and therefore its goals and objectives, within a predicted timeframe. Monitoring also provides critical information about whether a site requires maintenance or contingency actions. Monitoring is therefore essential for a project to achieve compliance.

The studies investigating whether compensatory wetland mitigation projects were required to be monitored and whether monitoring actually occurred are summarized in Table 6-8. In general, it appears that studies conducted more recently found that monitoring was required for a greater percentage of projects than was required in older projects. Data from four studies indicate monitoring was required for at least three-fourths of projects (Erwin 1991, Morgan and Roberts 1999, Johnson et al. 2000, Michigan Department of Environmental Quality 2000). The remaining two studies, which examined compensation projects permitted in the late 1980s and early 1990s, found that monitoring was required for a third to half of projects (Holland and Kentula 1992, Storm and Stellini 1994).

Less than half of the projects appeared to have been monitored. However, the studies did not determine whether the monitoring was never conducted or whether there was simply no record of the monitoring reports on file with the regulatory agencies. Since over half

of the studies mentioned difficulty finding complete project information from the agency files (Storm and Stellini 1994, Morgan and Roberts 1999, Johnson et al. 2000, Cole and Shaffer 2002), it is possible that monitoring reports may have been submitted to the appropriate agencies but the reports were lost due to a lack of follow-up and poor file maintenance. If monitoring is not conducted there is no means to trigger maintenance or contingency actions. The consequence of inadequate follow-up by regulatory agencies is discussed in Section 6.4.8.

Table 6-8. Percent of projects requiring monitoring and those actually monitored.

Location of Study and Reference No. [a]	% of Projects Requiring Monitoring	% of Projects Monitored
Washington (9)	71%	33%
Washington/ western (20)	53%	18%
California (7)	32%	NA
Michigan (15)	87%	35%
Pennsylvania (37)	NA	<10%
Tennessee (17)	89%	43%
Florida (5)	98%	38%[b] (62%)[c]

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

[b] Represents projects that were adequately monitored.

[c] Calculated from Erwin (1991) indicating all projects that received some level of monitoring.

6.4.7 Maintenance

Compensatory wetland mitigation sites require maintenance to help ensure that performance standards and goals will be achieved. Maintenance includes implementing corrective actions to rectify problems, such as an insufficient water supply or inappropriate water regime, invasive species infestation (e.g., reed canarygrass, bull frogs, tent caterpillars), trash, vandalism, or anything else that may result in non-compliance with permit requirements. Johnson et al. (2002) observed that a lack of maintenance was one of the main reasons for poor success of mitigation projects.

Results revealed that permitting agencies did not require all compensation projects to provide maintenance. Studies discovered that permits required site maintenance for 41 to 78% of projects (Erwin 1991, Storm and Stellini 1994, Michigan Department of Environmental Quality 2000). However, even fewer projects (20 to 60%) complied with their maintenance requirements (Erwin 1991, Michigan Department of Environmental Quality 2000).

The research did not investigate the reasons for low compliance with maintenance provisions. However, it may be assumed to be linked to inadequate monitoring or regulatory follow-up, or a lack of cooperation from the owner of the site.

6.4.8 Regulatory Follow-Up

Once compensatory wetland mitigation is required, it is the responsibility of the regulatory agencies to track the project over time and determine if it complies with permit requirements. A regulatory agency follows up on compensatory mitigation projects by:

- Ensuring that required monitoring reports are submitted on schedule
- Performing site visits to confirm monitoring results and attainment of performance standards
- Ensuring maintenance actions are undertaken on schedule
- Ensuring that appropriate contingency measures are enacted

Studies in Washington and Oregon indicated that about half of compensatory wetland mitigation projects received some regulatory follow-up in the form of site visits, phone calls, or letters (Kentula et al. 1992, Johnson et al. 2002). In Michigan only about a quarter of projects received any kind of follow-up after the permit was issued (Michigan Department of Environmental Quality 2000).

A few studies also examined the effect of regulatory follow-up on project compliance, success, or both. Robb (2002) alluded to the fact that the high number of non-compliant compensation projects resulted from a lack of follow-up and enforcement actions. In Washington a study noted that all of the projects lacking regulatory follow-up were either minimally or not successful, while two-thirds of the projects receiving some kind of follow-up were either fully or moderately successful (Johnson et al. 2002).

One team of researchers observed:

> *The most ecologically successful sites were generally those that had received follow-up work in the form of maintenance, replanting, or improvements to grading or water control structures in accordance with recommendations made by NJDEP [New Jersey Department of Environmental Protection] and other regulatory agencies after initial compliance inspections revealed problems* (Balzano et al. 2002).

Studies indicated that regulatory follow-up can help to ensure the effectiveness of compensation sites. It is assumed that applicants will be more likely to abide by monitoring requirements and submit monitoring reports if regulatory agencies are actively following up on projects. Since monitoring reports are meant to identify what is working and where there are shortfalls, maintenance actions can be initiated or contingency measures can be triggered to correct the shortfalls and problems as soon as possible. Therefore, follow-up may improve the compliance of compensation projects.

6.4.9 Summary of Key Points

- Most compensatory wetland mitigation projects were installed. However, compliance levels of compensatory mitigation projects were generally low due to shortfalls of wetland acreage, failure to achieve performance standards, and a lack of monitoring and maintenance.
- Two out of three studies found that more than half of projects were installed according to plan. Projects not installed to plan most often did not comply with grading and vegetation specifics. Although inconclusive, it may be that more recent projects are more frequently installed according to plans than projects in the 1980s.
- The majority of studies found that about 50% of projects achieved their required wetland acreage.
- Even if individual projects did not fully achieve their required acreage, most studies found that at least 66% of the overall required acreage of compensation had been established.
- It appears that more recent studies are providing performance standards, in addition to goals and objectives, as a tool for measuring compliance.
- The requirement for monitoring as a regulatory condition seems to be increasing in more recent studies (30 to 50% in the early 1980s; 75% in more recent studies).
- Over 50% of the studies noted that it was difficult to find complete project files, thereby making it difficult to document if monitoring was occurring or being tracked by regulatory staff.
- The research found that 41 to 78% of projects required maintenance; however, only 20 to 60% of projects complied with maintenance requirements.
- Studies in Washington and Oregon found that approximately half of projects received some regulatory follow-up.
- Two studies suggested that follow-up had a positive influence on the level of compliance and success for compensatory wetland mitigation projects.

How is compensatory wetland mitigation doing in Washington?*

Five studies of compensatory wetland mitigation have focused on projects in Washington State during the past decade. The studies examined success, ecological functioning, permit compliance, and achievement of required wetland area, though not all studies looked at the same factors in the same way. The results suggest that compensatory mitigation in Washington is neither fully successful nor completely unsuccessful.

Most studies found that less than half of wetland compensation projects are fully effective. In the most recent and comprehensive evaluation of compensation projects, Johnson et al. (2002) found that 13% of compensatory wetland mitigation projects were fully successful and 33% were moderately successful. In western Washington, Storm and Stellini (1994) determined that 24% of compensation projects functioned well. In King County, Mockler et al. (1998) indicated that 3% of projects replaced lost wetland functions (though the report provides no explanation for how this determination was made).

In terms of compliance, Johnson et al. (2000) determined that 29% of projects were in full compliance, while for King County Mockler et al. (1998) found that 21% of projects were meeting their required performance standards.

Kentula et al. (1992) examined Section 404 permit decisions for Washington from 1980 through 1986. Data indicated that permit decisions resulted in a wetland loss of 40 acres (16 ha). Johnson et al. (2002) determined that 24 acres (10 ha) of wetland were lost due to projects that did not successfully establish wetland area and the frequent use of existing wetlands for enhancement.

*Results have been simplified for this summary. Please refer to Sections 6.3 and 6.4 for complete information.

6.5 Types of Compensatory Mitigation

When discussing compensatory mitigation it is important to have a common understanding of the types of compensation that can be used to mitigate for wetland losses. This is difficult because various agencies and organizations often define types of compensation differently (Morgan and Roberts 1999). An added difficulty is that each type of compensation represents a range of activities on a continuum rather than a distinct procedure.

This section describes several types of compensatory mitigation:

- Restoration
- Creation

- Enhancement/exchange
- Preservation
- Mixed compensatory mitigation
- Wetland mitigation banking
- In-lieu fee programs

Definitions given to each of the mitigation types are discussed below, followed by a description of how frequently each type is used and its relative effectiveness.

6.5.1 Restoration

Of the types of compensation, restoration has the widest variety of definitions. The most general is the reestablishment of wetland conditions (i.e., area, functions, and values) at a location where they formerly existed but no longer exist (Johnson et al. 2000, Jones and Boyd 2000). Activities associated with this definition could include removing fill material, plugging ditches, or breaking drain tiles. Other definitions involve returning a site to some historic condition. Examples of these definitions include:

- Reestablishing historic hydrologic processes (National Research Council 2001) or hydrogeomorphic (HGM) classes (Johnson et al. 2000). Activities associated with this definition typically involve removing a levee or breaching a dike to reconnect an area to the floodplain or to tidal influence.
- "Return of an ecosystem to a close approximation of its condition prior to disturbance (NRC 1992). Restoration requires knowledge of the wetland type prior to disturbance and has the goal of returning the wetland to that type" (Gwin et al. 1999).
- Returning an altered wetland "to a previous, although altered condition (Lewis 1990)" (Gwin et al. 1999).
- "The process, or the result of the process of returning an area or ecosystem to some specific former condition" (Munro 1991).

Perhaps as a result of the numerous definitions, confusion about what constitutes restoration versus other types of compensatory mitigation can occur in regulatory permits and mitigation plans. The last three definitions in the list above could just as easily describe enhancement activities. For example, planting trees in a degraded wet pasture could be an attempt to return an ecosystem (the pasture) to an approximation of its prior condition (forested wetland).

In their study of compensatory mitigation projects in Tennessee, Morgan and Roberts (1999) mentioned that several projects were classified as restoration. Based on the activities specified, however, enhancement would have been a more appropriate term.

Similar confusion occurred between restoration and creation. As a result of this confusion, the effectiveness of restoration, as a type of compensation, is difficult to assess.

6.5.1.1 Use of Restoration

For compensatory mitigation, restoration is often cited as the highest priority or most recommended type of compensation "because it offers the highest probability of success (Krucznyski 1990, Kusler and Kentula 1990, USDA-SCS 1992)" (Morgan and Roberts 1999). However, it is unclear whether the "probability of success" is based on data or assumption.

This emphasis on restoration is not reflected in the number of freshwater, compensatory restoration projects implemented on the ground. Restoration tends to be one of the least utilized types of compensation (Jones and Boyd 2000). In fact, two studies mentioned that none of the projects involved restoration (Shaich and Franklin 1995, Gwin et al. 1999). Most of the studies that specifically mentioned the number or percentage of projects using a particular type of compensation found that 20 to 30% of projects involved some restoration of wetland acreage (Morgan and Roberts 1999, Johnson et al. 2000, Johnson et al. 2002). Projects employing restoration as the sole form of compensation are even fewer (Shaich and Franklin 1995, Johnson et al. 2000, Johnson et al. 2002).

In a departure from the other studies, Holland and Kentula (1992) found that 65% of permits required restoration. However, 42% of the compensatory wetlands they looked at were estuarine or marine. If estuarine and marine projects are subtracted, the percentage of restored, freshwater wetlands is similar to the other studies.

Morgan and Roberts (1999) suggest that the lack of compensatory wetland restoration projects is due to the fact that "most suitable restoration sites are 'prior converted' farmland and because sizable acreages are being restored under the Wetland Reserve Program . . . sites available for compensatory mitigation may be limited." In Washington, it is believed that restoration is not used very frequently because most wetland impacts are relatively small (less than 2 acres [0.8 ha]), and it is very difficult to find restoration opportunities for small sites that are not cost prohibitive. Restoration is typically most feasible and cost effective if done over a large area. In addition, some regulatory requirements, particularly for local governments, direct applicants to provide compensation on-site, which often precludes an opportunity for restoration.

6.5.1.2 Effectiveness of Restoration

While it is widely stated that restoration is the most effective approach, the data to substantiate this claim are sparse. Studies indicate that there is a limited use of restoration for compensatory mitigation in freshwater wetlands. Thus, there is a substantial lack of data with which to evaluate its effectiveness as a type of compensation.

In Washington Johnson et al. (2000) found that one of three restoration projects was in full compliance. Johnson et al. (2002) found that one of two restoration projects established the required acreage of wetland and was fully successful. In Florida, Erwin (1991) found that restoration successfully established 88 acres (36 ha) more wetland area than was required. The limited existing data appear to suggest that when wetlands are restored, they are relatively effective at compensating for permitted losses.

6.5.2 Creation

It is generally agreed that creation involves establishing wetland conditions (area, functions, and values) in a location where wetland conditions previously did not exist (Johnson et al. 2000) or "that was not a wetland in the recent past (within the last 100-200 years) (Kruczynski 1990, Lewis 1990)" (Gwin et al. 1999). "Typically, a wetland is created by excavation of upland soils to elevations that will support the growth of wetland species through the establishment of an appropriate hydroperiod (Kruczynski 1990, Lewis 1990)" (Gwin et al. 1999).

Gwin et al. (1999) made a distinction between creating a wetland that is isolated from existing wetlands (creation) and creating a wetland that is immediately adjacent to an existing wetland, thereby enlarging the existing wetland (expansion). No other studies made this distinction.

6.5.2.1 Use of Creation

Seven studies discussed how frequently creation was required as compensation. All noted that at least 30% and in some cases more than half of compensatory wetland projects were created or involved some creation (Holland and Kentula 1992, Shaich and Franklin 1995, Gwin et al. 1999, Morgan and Roberts 1999, Johnson et al. 2000, Jones and Boyd 2000, Johnson et al. 2002).

6.5.2.2 Effectiveness of Creation

In Washington, Johnson et al. (2000) found that 10% of created wetlands were in compliance. Seventy percent of creation projects established the required acreage of wetland, and 60% of created projects were either fully or moderately successful (Johnson et al. 2002).

In other states, however, created wetlands did not perform as well. Creation projects failed to establish 527 acres (213 ha) of required wetland area in Florida (Erwin 1991). In Tennessee, Morgan and Roberts (1999) found, "Most creation projects …were only partially successful because they failed to develop wetland characteristics throughout. . . . Problems with created wetlands were numerous and involved both site design and vegetation establishment."

The results on the effectiveness of creation are mixed. Though projects in Washington have poor compliance, other aspects of effectiveness are relatively good. However, other

states found poor effectiveness for created wetlands. The data therefore suggest that further study is warranted.

6.5.3 Enhancement/Exchange

Enhancement involves modifying a specific structural feature of an existing degraded wetland to improve one or more functions or values based on management objectives (Gwin et al. 1999, Johnson et al. 2000). Enhancement typically consists of:

- Planting vegetation
- Controlling non-native, invasive species
- Modifying site elevations or the proportion of open water to influence hydroperiods

Gwin et al. (1999) defined exchange as:

> *Enhancement taken to the extreme (Kruczynski 1990), with most or all of the wetland converted from one type to a different type. For example, resource managers may intend to enhance habitat value for waterfowl by excavating an area of open water within an existing emergent marsh. However, if the open water area replaces the emergent wetland or a large proportion of it, wetland types have been exchanged.*

Because enhancement involves altering an existing wetland to compensate for the loss of other wetlands, the scientific literature mentions three main concerns regarding its use:

- Enhancement fails to replace lost wetland area (Shaich and Franklin 1995, Morgan and Roberts 1999). For this reason, the state of Michigan does not allow the use of enhancement for compensatory mitigation (Michigan Department of Environmental Quality 2000).
- Enhancement may fail to replace wetland functions, since "a positive change in one wetland function may negatively affect other wetland functions (Kruczynski 1990, Lewis 1990)" (Gwin et al. 1999). In addition, "there commonly is disagreement about whether or not the practice implemented actually enhances conditions at a site" (Morgan and Roberts 1999).
- Enhancement may result in a conversion of HGM and/or Cowardin classes, typically producing a compensation wetland without natural analogues (Shaich and Franklin 1995, Gwin et al. 1999, Johnson et al. 2002). When enhancement is used for compensation in such cases:

  > *a single Section 404 decision results in the destruction of the wetland for which the permit was issued, along with the conversion of a second wetland to a different, often atypical, HGM type. This 'double whammy' means that exchange* [enhancement] *explicitly does not fulfill the objective*

of 'no-net-loss' of wetlands but, instead, ensures loss of wetland area, additional wetland disturbance, and changes in overall ecological function (Gwin et al. 1999).

6.5.3.1 Use of Enhancement

Studies indicated that more than one-third of compensation projects used enhancement of existing wetlands as compensatory mitigation (Shaich and Franklin 1995, Gwin et al. 1999, Morgan and Roberts 1999, Johnson et al. 2000, Johnson et al. 2002).

6.5.3.2 Effectiveness of Enhancement

The effectiveness of enhanced compensation wetlands was evaluated by only two studies, both conducted in Washington. The researchers found less than 13% of enhanced wetlands were in complete compliance, while 56% of enhanced wetland projects met the requirement for acreage of compensation. (For projects that proposed to enhance existing wetlands, establishing the required acreage of wetland compensation entailed implementing the proposed actions to enhance the mitigation site.) Furthermore, none of the enhanced compensation wetlands were fully successful, while 89% were minimally or not successful. (Johnson et al. 2000, 2002).

Johnson et al. (2002) suggested two main reasons for the low level of success among enhancement projects:

- The enhancement project did not achieve the proposed vegetative structure, diversity, or both (the planted trees and shrubs did not survive or did not grow). Thus the project did not establish the required acreage of compensation, did not meet performance standards, or both.
- The enhancement project achieved the proposed structure/diversity, but despite this, it did not adequately compensate for the wetlands lost because the contribution of the enhancement to the performance of wetland functions was low.

The enhanced wetlands evaluated by Johnson et al. (2002) were all in the ground for less than eight years. Their study confirmed that for the projects they evaluated, eight years was not sufficient time to achieve the structural and species complexity of shrub and forested habitats. When structurally complex habitats are the goal of a compensatory mitigation design, studies continue to show that longer timeframes are necessary to begin to provide some of the attributes of those functions. If structurally complex habitats are altered, the delay in replicating those functions results in a prolonged temporal loss of functions on the landscape. This is equally true for projects proposing to restore or create wetlands.

Wetland creation vs. enhancement: Which contributes greater functions?

Johnson et al. (2002) determined that created wetlands were significantly more successful than enhanced wetlands. These researchers assessed the potential of compensation wetlands to perform wetland functions. They then determined how much the activities associated with the type of compensation contributed to, or improved, the level of wetland function. For creation/restoration projects it was assumed that if wetland conditions were achieved then the compensation activities were responsible for providing the assessed level of wetland functions. Enhanced wetlands performed some wetland functions prior to implementation of compensation activities. The authors believed it was important to determine how much enhancement activities contributed to, or improved, the level of performance of functions at a compensation wetland. The authors believed this was particularly important since enhancement, as a compensation tool, is based on improvement of wetland functions.

The study compared the contribution of created sites and enhanced sites for three function categories. Results indicated that over half of the created sites provided high or moderate contributions to wildlife habitat, water quality, and water quantity functions. Over half of the enhanced wetlands provided minimal to no contribution to wetland functions. The vast majority of enhancement actions were targeted at improving wildlife habitat functions. However, the enhanced wetlands were typically surrounded by development and lacked the buffers and connectivity necessary to improve habitat for most wildlife. In addition, most of the wetlands that were enhanced already provided some water quality functions. Thus, creation of wetlands provided a significantly greater contribution to the performance of water quality functions than enhancement of wetlands. Contribution to wetland functions was one element of overall success.

It is important to note that many created wetlands and some enhanced wetlands result in Cowardin and HGM classes that are not typical for the landscape. This is discussed in more detail in Sections 6.8.2.2 and 6.8.5.1, respectively. Also, because enhancement provides less gain in function per acre than creation or restoration, replacement ratios are generally higher; refer to Section 6.6 for more information.

6.5.4 Preservation

Preservation means "the protection of an existing and well-functioning wetland from prospective future threats" (National Research Council 2001). Preservation, therefore, provides the opportunity to protect wetland areas that might otherwise be in jeopardy. Like enhancement, preservation does not produce any new wetland acreage; for that reason, some concerns have been raised regarding its use as compensation for permitted wetland loss:

- Preservation results in a net loss of wetland acreage.

- Preserved wetlands are generally not large enough to protect ecosystems and biodiversity over the long term (Whigham 1999).
- Preserved areas may not be checked by regulatory agencies to verify that they contain the specified acreage of wetland. For example, Morgan and Roberts (1999) observed that one of the larger preserved wetlands in their study was predominantly upland and "did not meet the criteria for being considered a jurisdictional wetland."

On the other hand, if an area can be verified as wetland, "Preservation of an existing wetland removes the uncertainty of success inherent in a wetland creation or restoration project and requires no construction to complete" (Washington State Department of Transportation 1999). Preservation, therefore, eliminates the risk of failure and temporal loss of wetland functions since the preserved area is already an existing wetland ecosystem.

6.5.4.1 Use of Preservation

The studies generally found that preservation was required as compensation for less than one-quarter of projects (Holland and Kentula 1992, Morgan and Roberts 1999, Johnson et al. 2000, Jones and Boyd 2000). Preservation generated about 2% of the compensatory wetland acreage in a study from San Francisco, California (Breaux and Serefiddin 1999). A report from the Washington State Department of Transportation (1999) indicated that 76% of state transportation departments in the United States use preservation as at least one component of compensatory mitigation and 38% use it as a stand-alone form of compensation.

6.5.4.2 Effectiveness of Preservation

There is a general lack of information about the effectiveness of preservation. Only one study examined the effectiveness of preservation as a type of compensatory mitigation. In Washington, Johnson et al. (2000) determined that all four of the projects involving preservation as the sole form of compensation were in compliance. Compliance for preservation projects entailed verifying that the area was preserved and free from development and that a deed restriction or conservation easement was in place to legally protect the parcel from future development.

6.5.5 Mixed Compensatory Mitigation

Mixed projects involve more than one type of compensatory mitigation. For example, a common proposal in the Pacific Northwest entails enhancing an existing wetland and creating additional wetland area immediately adjacent to it. Mockler et al. (1998) observed, "most sites consist of creation—a small pool graded for open water and emergents—and enhancement, typically of wetland buffer." Mixed compensation, however, can also occur on separate sites, such as a created wetland adjacent to the development site and a preserved wetland some distance away.

Several studies identified mixed compensation projects (Mockler et al. 1998, Gwin et al. 1999, Johnson et al. 2000, Johnson et al. 2002). For their studies of compensation wetlands, Mockler et al. (1998) and Johnson et al. (2002) classified compensation wetlands according to their dominant type of compensation. However, some projects lacked sufficient information to make this determination, while other projects lacked dominance by any one type of compensation.

6.5.5.1 Use of Mixed Compensation Projects

In the six studies that discussed how frequently mixed compensatory mitigation was required, results ranged from 13% (Johnson et al. 2002) to 43% (Johnson et al. 2000). Most studies found that mixtures were used for less than a third of projects (Holland and Kentula 1992, Shaich and Franklin 1995, Gwin et al. 1999, Morgan and Roberts 1999).

6.5.5.2 Effectiveness of Mixed Compensation Projects

Only two studies, both from Washington, examined the effectiveness of projects utilizing a mixture of compensation types. Johnson et al. (2000) found that 32% of mixed projects were in compliance. Johnson et al. (2002) determined that all of the mixed projects were moderately successful.

6.5.6 Wetland Mitigation Banking

Wetland banking provides an alternative for compensatory wetland mitigation that continues to grow in acceptance and use. It is defined as "the practice of restoring, creating, enhancing, or preserving off-site wetland areas to provide compensatory mitigation for authorized impacts to wetlands" (Environmental Law Institute 2002).

Typically a public agency, organization, or private entrepreneur establishes a bank on a large area to be used to compensate for a number of smaller wetland impacts. Generally, banks are established prior to the majority of wetland losses that they are meant to compensate for. This practice may provide advantages over traditional compensatory mitigation by reducing the temporal loss of wetland functions. For more information on wetland banking, refer to the Draft Programmatic EIS on *Washington State's Draft Rule on Wetland Mitigation Banking* (Driscoll and Granger 2001).

6.5.6.1 Use of Wetland Banking

Mitigation banking was required for about 7% of Section 404 permits in California issued from 1971 to 1987 (Holland and Kentula 1992). In the Norfolk District for permits issued from 1996 to 1998, about 10% of compensatory mitigation projects required the purchase of bank credits (Jones and Boyd 2000).

By the beginning of 1996, Brown and Lant (1999) determined that 68 banks had been established across the country, totaling nearly 41,000 acres (16,590 ha). A recent survey

by the Environmental Law Institute (2002) determined that 219 banks had been approved across 40 states, totaling more than 139,000 acres (56,250 ha). Though the majority of this acreage had not yet been used for compensation purposes, 22 banks had already sold all their eligible compensatory wetland acreage/credits (Environmental Law Institute 2002).

Since wetland bank credits result from one or more of the previously mentioned types of compensation, the Environmental Law Institute (2002) investigated how frequently each type was used in mitigation banking. Results indicated that 78% of banks involved multiple types of compensation and that enhancement and restoration are the most commonly used. Of the banks that relied on a single type of compensation, about a third was restoration; another third was creation, while enhancement and preservation were each used on 16% of the banks.

6.5.6.2 Effectiveness of Wetland Mitigation Banks

Only one study has examined the effectiveness of wetland mitigation banks. Brown and Lant (1999) examined banks that had been established by the beginning of 1996. Overall, they found there would be a net loss of over 21,000 acres (8,450 ha) of wetland due to the use of enhancement and preservation at banks. The authors also discovered that eight banks did not provide the functions required or specified, while four banks used or sold more acreage for compensation of wetland loss than was eligible from the bank (in other words, the bank was overdrawn).

Wetland mitigation banking is increasingly being used to compensate for wetland losses. Yet the only study investigating the effectiveness of banks raises concerns about its use. Further study will therefore be critical to determine the level of compliance and success of mitigation banks in providing functions.

6.5.7 In-Lieu Fee Programs

In-lieu fee programs provide an additional option for compensatory mitigation. They allow permit applicants to compensate for wetland losses by paying a fee to a third party such as a government agency or conservation organization (U.S. General Accounting Office 2001, Environmental Law Institute 2002). The fees are meant to be used to restore, create, enhance, or preserve wetlands (U.S. General Accounting Office 2001).

Generally, in-lieu fee contributions are collected in advance of wetland losses. These funds are accumulated until they are sufficient to design and implement the wetland compensation project (Environmental Law Institute 2002).

6.5.7.1 Use of In-Lieu Fee Programs

A recent survey by the Environmental Law Institute (2002) determined there were 87 active in-lieu fee programs across 27 states. “Through fiscal year 2000, developers used the in-lieu-fee option to fulfill mitigation requirements for over 1,440 acres [583 ha] of

adversely affected wetlands, and paid over $64.2 million to in-lieu-fee organizations" (U.S. General Accounting Office 2001).

6.5.7.2 Effectiveness of In-Lieu Fee Programs

Two studies discussed the effectiveness of in-lieu fee programs. However, neither study provided information on the level of compliance or ecological success of these programs.

A study by the U.S. General Accounting Office (2001) examined the effectiveness of in-lieu fee programs used by the Corps of Engineers to compensate for wetland losses permitted through Section 404. Of the 17 Corps districts using in-lieu fees, 65% did not require a specific timeframe for spending or obligating the fees received, and a few districts had not spent or obligated any funds though they had been collecting fees as compensation for wetland losses for at least three years (U.S. General Accounting Office 2001). The study found that three districts used the fees for research and/or education, rather than on-the-ground activities to compensate for wetland loss. In-lieu fee programs in 30% of the districts restored, enhanced, created, or preserved wetland acreage equal to or greater than the wetland acreage lost. The remaining districts either had used the fees to implement wetland activities that did not compensate for the wetland acreage lost, or they did not have any data (U.S. General Accounting Office 2001).

A study by the Environmental Law Institute found that 45% of in-lieu fee programs lacked the data necessary to determine their effectiveness. In-lieu fees replaced more wetland acreage than was lost in 56 programs, while "thirteen in-lieu-fee programs reported replacing fewer acres than had been impacted" (Environmental Law Institute 2002).

These studies paint a rather grim picture of the effectiveness of in-lieu fee programs as compensation for wetland loss. However, both in-lieu fees and mitigation banking can provide a mechanism to compensate for wetland impacts that currently do not require compensation, either because they are too small to be regulated on a federal, state, or local level or because they are considered non-jurisdictional (Shabman et al. 1993). In the year 2000 federal guidance on the use of in-lieu fee arrangements for compensatory mitigation was issued, while prior to this there were no federal requirements for in-lieu fee programs (Environmental Law Institute 2002). Further study will be needed to ensure that abuses of in-lieu fee compensation are not occurring.

6.5.8 Summary of Key Points

- The variety of definitions or criteria associated with types of compensatory mitigation has led to confusion in permitting and evaluating projects. For instance, comparing the effectiveness of one type of compensation with another is impossible when it is not clear if a project involved creation, restoration, enhancement, or some combination thereof.

- Restoration has been recommended as the "highest priority" method for compensation. Research in Washington has found that it is the least used, though one of three projects was in compliance and one of two projects was fully successful.

- Restoration may be used infrequently due to lack of site opportunity, or because of a preference by regulatory agencies and project applicants to provide compensatory mitigation on-site for development projects.

- Creation was used in one-third to one-half of compensation projects. In Washington, 10% of creation projects were in compliance, 60% were at least moderately successful, while studies from other states indicated that creation was less effective.

- Enhancement was used for compensation in more than one-third of compensation projects. Research in Washington found that less than 13% of enhancement projects were in compliance. There were no fully successful enhancement projects, while 89% were minimally or not successful.

- The low level of success for enhancement projects was attributed to an inability to achieve the proposed vegetative structure/diversity, a minimal gain in functions, or both. This may partially be a factor of time: There will be continued temporal loss of some functions until young sites mature to more complex structural conditions.

- Two studies from Washington indicated that mixed compensation projects had a higher level of compliance than creation or enhancement, and all mixed projects were moderately successful.

- Preservation can result in permanent protection of existing wetland resources, but compliance was found to be variable. One study found a large area of preserved wetland was actually predominantly upland habitat. However, a study in Washington found that 100% of preservation sites were in compliance.

- Studies of wetland mitigation banking and in-lieu fee programs focused on whether the goal of preventing the net loss of wetlands had been achieved on paper. The results indicated that a net loss of wetland area was occurring. A few banks were overdrawn, and some of the in-lieu fee programs had not used the money collected to implement compensation activities. No studies determined their effectiveness on the ground.

Does size influence the effectiveness of compensatory wetland mitigation projects?

Studies of the effect of wetland size on compensation projects revealed mixed results.

Two studies indicated that larger projects, which probably involved more planning and regulatory oversight, had a higher level of compliance (Brown and Veneman 2001) or success (Allen and Feddema 1996). Allen and Feddema (1996) noted that large projects (greater than 8.6 acres [3.5 ha]) resulted in a net gain of wetland acreage, while the smaller projects resulted in a net loss of wetland acreage. Though Brown and Veneman (2001) indicated larger projects had a higher level of compliance, larger projects were no more successful at replacing the plant communities or wildlife functions that were lost than the smaller compensation wetlands.

Two other studies determined that no significant correlation existed between wetland size and compliance or success (Balzano et al. 2002, Johnson et al. 2002). Raw data from Johnson et al. (2002) implied that compensatory mitigation projects 5 acres (2 ha) or larger were less successful than smaller projects. Balzano et al. (2002) found that larger compensation wetlands tended to be more successful at establishing the required wetland acreage. However, this trend was attributed to one large site (over 40 acres [16 ha]) that established more wetland acreage than was required.

The Committee on Mitigating Wetland Losses determined that wetland size does affect wetland functions (National Research Council 2001). For example, "for water quality purposes, many small wetlands would be more effective than one large wetland covering the same area." The committee therefore concluded that "replacement area should be proportional to the area required to replace the functions lost" (National Research Council 2001).

6.6 Replacement Ratios

A replacement ratio, or compensation ratio, is an approach used to determine appropriate reparation for permitted wetland losses, though not all regulatory agencies use this approach (for example, the U.S. Army Corps of Engineers considers the needs for compensation on a project-specific basis rather than assigning replacement ratios).

The replacement ratio reflects the acreage of a particular type of compensatory mitigation (creation, restoration, enhancement, or preservation) needed to make up for the loss of an acre of wetland (King et al. 1993, McMillan 1998). For example, a permitted loss of 1 acre may have to be compensated for with 6 acres of enhancement, thus requiring a 6:1 replacement ratio.

This section provides the following information:

- The rationale for using replacement ratios
- A summary of the literature regarding what replacement ratios are being required and if they are being achieved
- Some of the methods used to determine appropriate replacement ratios

6.6.1 Rationale for the Use of Replacement Ratios

When compensatory wetland mitigation was first required, the loss of an acre of wetland would simply require an acre of compensation (McMillan 1998). A simple 1:1 replacement ratio generally is no longer considered appropriate (Castelle et al. 1992, King et al. 1993, National Research Council 2001) for the following reasons:

- **Risk of failure.** It is possible that compensation projects will not perform as proposed (King and Bohlen 1994) and may fail to compensate for wetland losses (Castelle et al. 1992).
- **Temporal loss.** It may take anywhere from several years to several decades for a compensation project to achieve ecological equivalency (National Research Council 2001) and to develop the proposed/required wetland structures and/or functions (Castelle et al. 1992).

Because of the risk of failure and temporal loss, "replacement ratios greater than 1:1 are used as a means of equalizing the tradeoff. While the goal is always to replace the lost functions at a 1:1 ratio, it is almost always necessary to increase the replacement acreage in order to accomplish this" (McMillan 1998).

A literature review performed by Castelle et al. (1992) concluded that:

> *the risks of project failure and the time it takes for a created wetland to represent a fully functioning ecosystem should be factored into replacement ratios which exceed 1:1. Replacement ratios of 2:1 or greater are necessary to compensate for our current rate of failure to achieve permit compliance of basic wetland community structural objectives within attempted mitigation projects, neither of which are accurate measures of functional equivalency.*

An additional consideration is that there are many types of wetlands and various degrees of degradation. As a result, not all wetlands provide the same levels of functions or values. Replacement ratios, therefore, must take into account the type and quality of the wetland and the functions and values that would be lost. For example, the loss of a high-quality forested wetland would require a higher replacement ratio than the loss of a highly degraded wet pasture (Breaux and Serefiddin 1999).

Also, the type of compensation can influence the replacement ratio. For example, the use of enhancement results in a net loss of wetland area and a limited increase in wetland functions (Johnson et al. 2002). Thus, enhancement typically requires higher replacement ratios than restoration or creation (McMillan 1998).

Higher replacement ratios result in more area for compensatory mitigation, but unfortunately size does not guarantee success or quality. A study conducted by the National Academy of Sciences concluded that attempts to compensate for rare wetland types by requiring high replacement ratios yielded wetlands of a common type at a low ratio. Rather than replicating the rare wetland type, a more common wetland type was substituted. "In effect, the regulatory program may reassemble the landscape with a different habitat mix than the wetlands being lost" (National Research Council 2001).

6.6.2 Replacement Ratios Required and Achieved

Table 6-9 summarizes the overall or average replacement ratios that were required for compensatory wetland mitigation projects. A wide range of replacement ratios was required—from 0.66:1 to 5.9:1 (Kunz et al. 1988, Johnson et al. 2000). These are the extremes. The low end represents projects from the early to mid 1980s, when compensatory mitigation was still a relatively new idea. The higher ratios reflect more recent projects using predominantly enhancement and/or preservation, which typically require higher replacement ratios.

Between these extremes, the remaining studies noted required ratios ranging from 1.5:1 to 2.7:1 (Wilson and Mitsch 1996, Morgan and Roberts 1999, Michigan Department of Environmental Quality 2000, Balzano et al. 2002, Johnson et al. 2002, Robb 2002).

Actual replacement ratios that were achieved for the projects studied are also shown in Table 6-9. None of the studies found the required ratios had been realized. In fact, Balzano et al. (2002) determined that forested compensation wetlands achieved only 1/100th of an acre for every acre lost despite the fact that over 2 acres of forested wetland were required. Achieved ratios ranged from 0.7:1 to 1.9:1 (Wilson and Mitsch 1996, Morgan and Roberts 1999, Balzano et al. 2002, Johnson et al. 2002, Robb 2002).

As mentioned in the previous section, replacement ratios typically require greater than 1:1 replacement to factor in the risk of failure. Table 6-9 demonstrates the utility of this approach since all of the studies indicated that the achieved ratios were smaller than those required. All but one of the studies found the achieved ratios were greater than 1:1, though not by a significant margin. But two of these studies included enhancement of existing wetlands.

Ratios are a tool to address the temporal loss of wetland functions and the historic failure of replicating wetland acreage and functions. The results indicate an inability of compensation projects to achieve their required replacement ratios. This inability reflects the same problems and shortfalls associated with compensation project success and compliance (see Section 6.4).

Table 6-9. Comparison of replacement ratios that were required and achieved.

Location of Study and Reference No. [a]	No. of Projects Evaluated	Replacement Ratio Required	Replacement Ratio Achieved	Comments
Washington (10)	24	2.2:1 [b]	1.87:1 [b]	Enhancement accounted for 65% of the established acreage
Washington (9)	45	5.9:1 [b]	NA	Acreage predominantly preservation and enhancement
Washington (13)	35	0.66:1	NA	Corps and EPA data 1980 to 1986
Michigan (15)	76	1.82:1 (average)	NA	Required ratios ranged <1:1 to >5:1; study did not include enhancement
Indiana (18)	31	2.5:1	1.1:1	Achieved ratios for specific Cowardin classes ranged from 0.48:1 for PFO to 45:1 for POW. Study did not include enhancement
Ohio (22)	4 (5)	1.5:1 (1.7:1) [b]	1.4:1 (0.7:1) [b]	Study reviewed 5 projects, results and conclusions focus on 4; parentheses reflect results for all 5
New Jersey (2)	75	1.8:1 (average) 2.04:1 PFO 2.78:1 PSS 1.85:1 PEM 1.07:1 POW	0.78:1 (average) 0.01:1 PFO 0.91:1 PSS 1.29:1 PEM 0.28:1 POW	Sites proposing POW did not achieve the required acreage. However, POW was on sites that did not propose to have open water; thereby resulting in three times more POW acreage than required
Tennessee (17)	47	2.7:1	1.9:1	Ratio = 0.88:1when enhancement and preservation are excluded

NA = not available

PFO = palustrine forested; PSS = palustrine scrub-shrub; PEM = palustrine emergent; POW = palustrine open water

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

[b] Calculated from data provided.

6.6.3 Approaches for Determining Replacement Ratios

King et al. (1993) proposed a framework for calculating replacement ratios, "based on the idea that compensatory mitigation involves trading one form of environmental capital for another and that full compensation requires increases in environmental functions and values from the compensation wetland that are sufficient to make up for the decline in functions and values resulting from the loss of existing wetland." The authors mentioned five parameters to consider when determining an appropriate replacement ratio:

- The pre-existing level of wetland function per acre at the site proposed for wetland compensation. In the case of enhancement, this ensures that an applicant

does not get mitigation credit for functions that were already being provided by a pre-existing wetland.

- The maximum level of function anticipated to be provided by the wetland compensation project.
- The number of years after construction that will be required for the wetland compensation project to reach its anticipated or sustainable level of function.
- The number of years between the loss of the original wetland and the completion of construction of the compensation wetland (temporal loss). Mitigation could be done concurrent with impacts, in advance, or delayed after impacts occur.
- The likelihood that the project will not achieve its anticipated level of function.

King et al. (1993) suggested entering the values for each of the five parameters into an analytic model that then calculates an appropriate compensation ratio for the project-specific information provided. King and Bohlen (1994) provided easy to use tables of replacement ratios that would result from a variety of values for the five parameters identified above.

Using parameters comparable to King et al. (1993) for determining appropriate compensation, Rheinhardt et al. (1997) described an approach based on function assessment. The authors proposed the following steps:

1. Develop a function assessment method for the specific regional conditions, including identification of reference wetlands.
2. Assess wetlands proposed to be lost, thereby determining the level of each wetland function that will be lost.
3. Assess potential compensatory mitigation sites to evaluate their current level of function and predict future conditions and levels of function that would result from mitigation activities within the timeframe required for regulatory monitoring.
4. Calculate ratios for compensation for each function "by dividing the degree to which a function is reduced through project alteration by the degree to which a function is increased through restoration" (Rheinhardt et al. 1997).

The ratio "varies among functions and is influenced by (1) the magnitude to which any given function occurs at a project site both before and after the site is altered, (2) the magnitude to which any given function occurs at a compensatory mitigation site both before and after restoration is applied, and (3) the rate at which any given function is restored" (Rheinhardt et al. 1997). The goal of this approach is not just to ensure no net loss of wetland functions but also to restore wetland ecosystems (Rheinhardt et al. 1997).

In contrast, Breaux and Serefiddin (1999) argue that "there has not been any single, universally accepted assessment procedure to determine wetland functions and values

(Kusler 1997)." As a result they suggest that "the quantitative measure of area provides a degree of certitude that should be taken advantage of" (Breaux and Serefiddin 1999). In other words, assessing or determining the level of functions provided by a wetland can be time-consuming to near impossible, while acreage provides an easy measurement. The authors mention a few conditions that may require greater than 1:1 replacement ratios (for example 2:1 or 3:1):

> *If it is determined that the area lost includes functions and values of high quality…*
>
> *If . . . the replacement area is outside the watershed, sub-watershed, or county;*
>
> *If the replacement area involves a high risk of failure or uncertain outcome,*
>
> *If there are high temporal losses . . .*
>
> *If the habitat loss is likely to be substantially greater than the creation of new habitat; or*
>
> *If the connection between two wetland sites is severed or a large site is divided.* (Breaux and Serefiddin 1999).

Robb (2002) also proposed using an acreage-based system for determining appropriate replacement ratios. However, where the previous studies did not base ratios on the type of wetland, his system focused on developing replacement ratios for each Cowardin class. This approach resulted from a delineation of 31 compensatory mitigation sites in Indiana. Robb (2002) compared the required acreage of each Cowardin class with the acreage that was established. For example, results indicated that 71% of the required acreage of palustrine forested wetlands was not established (a 71% rate of failure). The ratio recommended to overcome this failure was calculated by dividing the required acreage by the acreage actually established. Using this data, the ratio for palustrine forested wetlands should therefore be 3.5 acres of compensation for every acre of wetland lost. The rationale was that for every 3.5 acres constructed that were intended to be palustrine forested wetland, 1 acre would actually become forested wetland. Proposed ratios for other wetland types included:

- 1.8:1 for scrub-shrub
- 7.6:1 for wet meadow
- 1.2:1 for shallow marsh
- 1:1 for open water

Robb (2002) conceded that his study did not consider the quality of the compensation wetlands or whether they replaced the functions lost. The author mentioned that more

regulatory follow-up could result in more successful projects and therefore lower replacement ratios.

6.6.4 Summary of Key Points

- Replacement ratios provide a means of taking into account the potential failure and temporal loss of functions as well as the potential gain in acreage or functions to be provided by the compensation project.
- Several methods are available to calculate replacement ratios on a case-by-case basis. Examples of some of the criteria used to determine ratios include the functions proposed to be provided at the compensation site, the functions anticipated to be lost at the impact site, size, landscape position, and relative chance of success.
- Required replacement ratios vary from one state to another, based on the type of compensation proposed, and based on project-specific circumstances.
- Studies found that compensation projects did not achieve their required replacement ratios. In some cases this resulted in less than 1:1 acreage replacement.

6.7 Replacement of Wetland Acreage

This section summarizes the results of studies examining whether compensatory wetland mitigation is replacing the acreage of authorized wetland losses. Replacement of wetland acreage is similar to "no net loss," which refers to a goal for the nation and Washington State to ensure there will be no overall net loss in acreage and function of the remaining wetland resource base (The Conservation Foundation 1988, McMillan 1998). The no-net-loss goal, however, "does not mean that no further wetlands will be lost; rather, that mitigation and non-regulatory restoration will offset wetland losses" (McMillan 1998). Replacement of wetland acreage, on the other hand, focuses on wetland losses and gains associated with compensatory wetland mitigation.

Replacement of wetland acreage provides a measurable and consistent method for evaluating and comparing the effectiveness of compensatory mitigation programs (Kusler 1988). The scientific literature contained two types of information on this topic:

- Studies that evaluated how well permitting programs (e.g., Section 404) achieved replacement of wetland acreage. Most of these studies used information from permit files and databases.
- Studies that evaluated how well compensation projects achieved the replacement of wetland acreage on the ground. These studies were conducted in the field and typically involved wetland delineations.

6.7.1 Programmatic Evaluations of Acreage Replacement

Programmatic evaluations, in contrast to most of the studies mentioned thus far, are not concerned with the effectiveness of individual compensatory mitigation projects. Instead, programmatic evaluations focus on whether a permitting agency or permit program is requiring sufficient wetland acreage compensation to replace the authorized wetland losses occurring over a specified time.

In a programmatic evaluation, wetland acreage replacement is determined by comparing the acreage of wetlands lost, or adversely altered, with the acreage of wetlands required for compensatory mitigation in a specific geographic area. These evaluations typically rely on information from permit files and databases, rather than verification of on-the-ground, as-built conditions.

Five studies examined the effectiveness of wetland permitting and compensatory mitigation programs (Table 6-10). The earliest study reviewed Section 404 permit data from Washington, 1980 to 1986, and Oregon, 1977 to 1987, "to describe how permit decisions affect the wetland resource" (Kentula et al. 1992). Results indicated that in Washington 39 acres (16 ha) of wetland were not replaced, while in Oregon 79 acres (32 ha) of wetland were not replaced. The authors also observed, "In Washington, approximately 3 percent of the permits issued required compensatory mitigation" (Kentula et al. 1992).

The results of this study should be considered within the context of the Seattle District Corps of Engineers regulatory program in the early 1980s. Compensatory mitigation, when it was required, was only required for projects that triggered an individual permit. In the early 1980s, the threshold for wetland fill under Nationwide Permit 26 was 10 acres (4 ha). Fill of 10 acres or less in isolated wetlands was permitted outright. Therefore, the 39 acres of wetland identified by Kentula (1992) as "not replaced" very likely represents but a fraction of the total acreage of permitted wetland losses that were not compensated for at that time.

A study of Section 404 permitting from southern California noted that 8 acres (3 ha) of wetland were not replaced (Allen and Feddema 1996). The study also determined that "freshwater wetlands are experiencing a disproportionately greater loss of area and that riparian woodland wetlands are most often used in mitigation efforts. The net result of these accumulated actions is an overall substitution of wetland types throughout the region" (Allen and Feddema 1996).

Two of the remaining studies generally found that permitting programs required a net gain from compensatory mitigation (Table 6-10). Gains in acreage ranged from about 47 acres (19 ha) (Torok et al. 1996) to nearly 197 acres (80 ha) (Holland and Kentula 1992). However, the study of the effectiveness of the New Jersey Freshwater Wetlands Protection Act (Torok et al. 1996) mentioned compensatory mitigation acreage only for individual permits. It was not clear from the article if any of the 3,003 general permits, resulting in over 600 acres (243 ha) of wetland loss, required any compensatory mitigation. Furthermore, Holland and Kentula (1992), in their evaluation of Section 404

permitting in California, noted that data on acreage of impacts and compensation were lacking in about 40% of the permit files.

The fifth study focused on the Norfolk Corps District (Jones and Boyd 2000). The authors indicated that new wetland acreage produced by creation or restoration did not fully replace the permitted wetland losses, thereby resulting in a loss of about 260 acres (105 ha) (Jones and Boyd 2000). However, preservation, mitigation bank credits, and substantial in-lieu fee contributions provided additional compensation. If acreages from all types of compensatory mitigation are included, the authors assumed there was a gain of at least 1,500 acres (607 ha). Despite the fact that only 24% of the permits required compensation, the authors concluded that replacement of wetland acreage was achieved, at least on paper (Jones and Boyd 2000).

The results (Table 6-10) appear to indicate that since the early 1980s, permitting programs have required an increasing amount of acreage to compensate for wetland losses. Whereas permits from the mid-1980s did not appear to require the replacement of acreage for wetland losses, permits from the mid- to late 1990s did appear to require replacement of wetland acreage.

Table 6-10. Permitted wetland loss compared to required wetland compensation.

Location of Study and Reference No. [a]	No. of Permits	Wetland Area Lost	Area of Compensation Required	Comments
Washington (12)	35	152 acres (61.4 ha)	112 acres (45.5 ha) created	Section 404 permits 1980-1986
Oregon (12)	58	183 acres (73.9 ha)	103 acres (41.8 ha) created	Section 404 permits 1977-1987
California (7)	324	2,907 acres (1,176.3) ha	3,103 acres (1,255.9 ha)	Section 404 permits 1971-1987; data on acreages was often lacking
California/ southern (1)	75	199 acres (80.5 ha)	191 acres (77.3 ha) completed	Section 404 permits 1987-1989; permits required 276 acres (111.6 ha) of compensatory mitigation
Norfolk Corps District (11)	1692	863.8 acres (349.6 ha)	538.6 acres created (218.0 ha) 65.5 acres restored (26.5 ha) 1,537.2 acres preserved (622.1 ha) 200.8 bank credits $2,574,966 in lieu fee	Section 404 permits 1996-1998

Location of Study and Reference No. [a]	No. of Permits	Wetland Area Lost	Area of Compensation Required	Comments
New Jersey (21)	3003 (107)	602 acres (243.8 ha) *164 acres (66.5 ha)*	NA *171 acres (69.2 ha) created;* *41 acres (16.5 ha) restored*	New Jersey Freshwater Wetlands Protection Act permits 1988-1993. Numbers in italics are individual permits; all other numbers are state general permits.
[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.				

6.7.2 Project-Specific Evaluations of Acreage Replacement

Studies that examined the effectiveness of compensation projects often assessed whether the projects achieved replacement of wetland acreage. The assessment generally involved determining how much wetland acreage the compensation projects provided. The wetland compensation acreage produced on the ground was then compared to the acreage of wetland loss associated with those projects. If the compensation acreage was less than the wetland acreage lost, a net loss of wetland occurred. Seven studies analyzed compensatory wetland mitigation project data to determine whether replacement of wetland acreage was achieved.

Four studies either focused on creation or restoration, or they did not mention the type of compensation. The studies noted that the acreage of wetland compensation was less than the acreage of wetland loss by as much as 34%, thereby resulting in a net loss of up to 8 acres (3 ha) (Gwin and Kentula 1990, Allen and Feddema 1996, Wilson and Mitsch 1996). However, a study conducted for the South Florida Water Management District found that creation and restoration activities resulted in 106% of the wetland acreage lost—a net gain of almost 65 acres (26 ha) of wetlands (Erwin 1991).

One issue that emerges when considering replacement of wetland acreage is the use of enhancement and preservation as wetland compensation. Three studies noted that enhanced or preserved wetlands accounted for 45 to 65% of the acreage of compensation (Shaich and Franklin 1995, Morgan and Roberts 1999, Johnson et al. 2002). In Washington nearly two-thirds of the established acreage of compensation involved enhancing existing wetlands, while creation and restoration of wetland area replaced only 65% of the permitted wetland losses (Johnson et al. 2002).

Some authors discounted the acreage provided by enhancement and preservation. Enhancement and preservation are often not included in determining net loss or gain because neither type of compensatory mitigation produces any new wetland acreage (Breaux and Sereffidin 1999). When acreage provided by enhancement and preservation are disregarded, three studies found wetland losses of 22, 11, and 24 acres (9, 4, and 10 ha) respectively (Shaich and Franklin 1995, Morgan and Roberts 1999, Johnson et al. 2002). This equaled 58, 12, and 41% of the authorized wetland losses, respectively (Shaich and Franklin 1995, Morgan and Roberts 1999, Johnson et al. 2002).

6.7.3 Summary of Key Points

- Programmatic evaluations have documented an increase in the acreages of wetland compensation required since the early 1980s. However, the acreage of wetland replacement may include preservation, enhancement, or both.
- Project-specific data revealed that compensation wetlands did not replace the acreage of wetlands that were lost. Even larger losses occurred if the acreages of enhancement and preservation were discounted.

6.8 Functions and Characteristics Provided by Created, Restored, or Enhanced Wetlands

This section describes the functions and characteristics provided by wetlands created, restored, or enhanced for compensatory mitigation and non-regulatory projects. It includes:

- The capacity of created and restored wetlands to provide wildlife habitat for invertebrates, amphibians, and birds. Wildlife habitat was evaluated through direct observations or evidence of wildlife use, the presence of structural indicators, or comparison to reference wetlands.
- The ability of created, restored, or enhanced wetlands to develop plant communities and vegetative characteristics. Studies involved comparisons with reference wetlands and investigations of factors affecting vegetation.
- The importance of soil conditions, particularly as they relate to establishing vegetation and improving water quality. Soil properties of created and restored wetlands were compared with reference wetlands.
- The ability of created and restored wetlands to provide water quality functions.
- The importance of water regime and how the creation and enhancement of compensation wetlands can result in atypical water regimes.

The scientific literature indicated that the ability of compensatory wetland mitigation projects to perform wetland functions is not noticeably different from that of non-regulatory restoration or creation projects. Newly implemented wetland sites face similar challenges and develop in similar ways regardless of whether they were legally required or voluntarily initiated.

Refer to Chapter 2 of this document for a discussion of the functions that wetlands provide.

6.8.1 Wildlife Habitat

Most articles focused on the ability of a created or restored wetland to provide habitat for one specific guild or group of animals, such as invertebrates, amphibians, or birds. Information on other habitat functions provided by created or restored wetlands was lacking.

6.8.1.1 Invertebrates

Several studies have compared invertebrate communities of created or restored wetlands with those of reference wetlands. Most of these determined that reference wetlands were more diverse, had greater taxon richness, or had higher density of species than created or restored sites (Brown et al. 1997, McIntosh et al. 1999, Fairchild et al. 2000, Dodson and Lillie 2001). One study, however, found "no convincing differences" in fly (dipteran) densities between created and reference wetlands (Streever et al. 1996). None of these studies were conducted in the Pacific Northwest. However, the results should be broadly applicable to wetlands anywhere.

The age of the wetland, or the amount of time elapsed since restoration occurred, was an important factor influencing invertebrate taxon richness, abundance, and/or diversity (Brown et al. 1997, Fairchild et al. 2000, Dodson and Lillie 2001). For example, "insects with aerial dispersal capability rapidly colonized the restored habitats, but some less mobile forms (non-insects and some hemipterans [true bugs]) either colonized more slowly or not at all" (Brown et al. 1997). Dodson and Lillie (2001) determined that a newly restored site would require 6.4 years for the zooplankton taxon richness to resemble that of a minimally disturbed reference wetland.

The growth and development of vegetation also appears to affect invertebrate communities (Chovanec 1994, Brown et al. 1997, Chovanec and Raab 1997, McIntosh et al. 1999, Fairchild et al. 2000). For example, certain predatory groups of beetles were early colonists at young sites with limited development of vegetation, while herbivorous beetle groups occurred at older sites after specific types of vegetation had developed (Fairchild et al. 2000). McIntosh et al. (1999) concluded, "wetlands at different successional stages may contain very distinct aquatic macroinvertebrate assemblages which may be important to the food web and other functional processes of wetlands."

6.8.1.2 Amphibians

The amphibian habitat present in created or restored wetlands has been compared with that of reference wetlands in several studies. On the east slope of the Cascade Range in the Teanaway and lower Swauk River drainages of Kittitas County, Quinn et al. (2001) found no difference in species richness of amphibians between created and reference wetlands, "although sample sizes may have been too small for differential species-use patterns to emerge." Other authors determined that created and restored wetlands differed from reference wetlands in terms of amphibian community structure, species

richness, or stomach content (Bursey 1998, Lehtinen and Galatowitsch 2001, Pechmann et al. 2001).

Though created or restored wetlands provide habitat for some amphibian species, conditions within the wetland and conditions outside the wetland may limit productivity, dispersal, colonization, or all three. Conditions that appear to affect amphibian communities include hydroperiod, presence of emergent vegetation, presence of fish, substrate, and availability of an invertebrate prey source (Bursey 1998, Baker and Halliday 1999, Monellow and Wright 1999, Pechmann et al. 2001).

Distance to other wetlands, connectivity between habitats, and the land use of the surrounding terrestrial habitats affect amphibian communities outside of a restored or created wetland (Baker and Halliday 1999, Monellow and Wright 1999, Lehtinen and Galatowitsch 2001, Pechmann et al. 2001). For example, Baker and Halliday (1999) observed that two species of amphibians dispersed to new ponds only if they were within 1,312 feet (400 m) of an existing pond, while two other species colonized new ponds up to 3,117 feet (950 m) from an existing pond.

Monellow and Wright (1999) concluded, "The interconnectiveness of amphibian habitat is an essential element in sustaining amphibian populations because it allows amphibians to overcome large population fluctuations and recolonize areas where populations have been extirpated."

Lehtinen and Galatowitsch (2001) found that the wetlands restored in urban areas had the lowest amphibian species richness. However, another study observed seven amphibian species, with as many as six species breeding, in created wetlands located in an intensively used urban recreational area near Vienna, Austria (Chovanec 1994).

6.8.1.3 Birds/Waterfowl

All the studies that examined the ability of created or restored wetlands to provide habitat for birds focused on non-regulatory projects. Therefore this section does not contain information on the ability of compensatory wetland mitigation projects to provide habitat for birds. However, the information is still relevant based on the similarity of results among compensatory and non-regulatory projects for the other studies of functions. None of the studies cited below were conducted in the Pacific Northwest.

Studies comparing bird use of created or restored wetlands and reference wetlands demonstrated variable results, perhaps indicating that site-specific conditions influence bird use. For example, two studies found no difference in bird abundance between restored and reference wetlands (Brown and Smith 1998, Ratti et al. 2001), while two other studies determined that reference wetlands had greater bird species richness and abundance (Delphey and Dinsmore 1993, Dobkin, et al. 1998). Though Brown and Smith (1998) found no difference in bird abundance, they did observe that bird density was greater at reference wetlands. Regardless of the findings for bird populations in general, two studies noted that ducks had similar or greater abundance, species richness,

or density at created and restored wetlands (Delphey and Dinsmore 1993, Ratti et al. 2001).

In the literature, the main factors that appeared to affect wetland use by bird populations were:

- The percent cover of emergent vegetation (Belanger and Couture 1988, Hemesath and Dinsmore 1993, VanRees-Siewert and Dinsmore 1996)
- The density and abundance of invertebrates (Belanger and Couture 1988, Cooper and Anderson 1996)

Though the age of the wetland did not directly affect overall bird populations at created and restored wetlands, VanRees-Siewert and Dinsmore (1996) noted that the richness of breeding bird species was significantly greater at older restoration sites. The composition of the bird community changed with age. Both of these effects were associated with an increase in the emergent vegetation in older wetlands.

6.8.2 Plants

6.8.2.1 Comparisons with Reference Wetlands

This section discusses studies that compared the vegetation of created and restored wetlands to that of reference wetlands. The studies examined a variety of parameters in a number of states and found variable results. Only one study determined that there was no difference in vegetation between created/restored and reference wetlands (Brown 1991). Two studies were conducted in the Pacific Northwest.

In the metropolitan area of Portland, Oregon, reference wetlands differed significantly from mitigation wetlands in terms of floristic composition. Mitigation wetlands had higher overall plant species richness, higher average percentage of native species, and significantly higher average occurrence of introduced and invasive/introduced species than reference wetlands (Magee et al. 1999).

Another study was conducted in the northwestern Great Basin on land that had previously been grazed by cattle. The study found that "sedge cover, forb cover, and foliage height diversity of herbs were greater" on reference plots, in which livestock had been excluded for more than 30 years. "[B]are ground, litter cover, shrub cover, and shrub foliage height diversity were greater" on restored plots, in which livestock grazing pressure had been removed prior to commencement of the study (Dobkin et al. 1998). During the four-year study period, restored plots experienced an increase in grass, forb, rush, and cryptogamic cover, but sedge cover did not change. The authors concluded, "the lack of change in sedge and shrub cover on open [restored] plots suggests that restoration to a sedge-dominated meadow will not happen quickly" (Dobkin et al. 1998).

Restored prairie pothole wetlands were found to lack low prairie and wet meadow zones that reference wetlands possessed. Restored wetlands had significantly higher richness of

submersed aquatics and greater coverage by mudflat and open water (Delphey and Dinsmore 1993, Galatowitsch and van der Valk 1995, Galatowitsch and van der Valk 1996). The researchers concluded that restored wetlands are not likely to develop the sedge meadow and wet prairie zones present in reference wetlands (Galatowitsch and van der Valk 1995, VanRees-Siewert and Dinsmore 1996).

Other authors determined that reference wetlands exhibited greater percent cover by wetland species. However, created wetlands had species richness that was equal to or greater than reference wetlands (Moore et al. 1999). Restored wetlands had significantly lower wetland index values (indicating that wetland species were providing more of the total vegetative cover) than reference sites (Brown 1999).

The common finding that created and restored wetlands have greater vegetation species richness is probably linked to the level of disturbance associated with creation and restoration and the broad range of niches created on a new site. For example, a newly created or restored site is like a *tabula rasa* (a blank slate) upon which species will be planted (installed or seeded), species from the previous habitat on the site will re-emerge, and species adapted to disturbance will colonize.

6.8.2.2 Cowardin Classes Provided by Compensatory Mitigation Wetlands

Cowardin class refers to a method used to categorize wetlands based on the dominant type of vegetation (Cowardin et al. 1979), as well as other factors. The main Cowardin classes used to categorize freshwater wetlands are:

- Emergent
- Scrub-shrub
- Forested
- Aquatic bed
- Open water (though not technically a Cowardin class, open water is often used to map and describe unvegetated areas of inundation)

Several studies evaluated compensatory wetland mitigation sites to determine which Cowardin classes were being established. Nearly all of these studies found that compensatory mitigation resulted in more acreage of open water/aquatic bed/deep marsh than was originally lost or required (Kentula et al. 1992, Shaich and Franklin 1995, Bishel-Machung et al. 1996, Magee et al. 1999, Cole and Brooks 2000, Michigan Department of Environmental Quality 2000, Balzano et al. 2002, Johnson et al. 2002, Robb 2002).

For example, in Washington State over 16 acres (6 ha) of open water/aquatic bed wetlands were gained (Johnson et al. 2002). In the Portland metropolitan area of Oregon, 29 acres (12 ha) of open water were gained (Shaich and Franklin 1995), and Indiana

gained over 3 acres (1 ha) of open water/deep marsh/aquatic bed (Robb 2002). Compensatory wetland mitigation projects in New Jersey generated 50 acres (20 ha) more open water than was required (Balzano et al. 2002).

Results for other Cowardin classes were more variable. For example, four studies noted either a loss of forested wetland area (4 to 8 acres [2 to 3 ha]) or an inability to establish this wetland class (Shaich and Franklin 1995, Bishel-Machung et al. 1996, Brown and Veneman 2001, Balzano et al. 2002, Robb 2002). On the other hand, a study from Washington State observed a net gain of over 12 acres (5 ha) in forested/scrub-shrub wetlands (Johnson et al. 2002).

Additional variability occurred in the balance of emergent wetlands. Two studies from the Pacific Northwest noted a loss of 35 to 51 acres (14 to 21 ha) for emergent wetlands due to their conversion to other Cowardin classes (Shaich and Franklin 1995, Johnson et al. 2002). Studies from other states, meanwhile, found that emergent wetlands were established more successfully than other wetland classes (Bishel-Machung et al. 1996, Brown and Veneman 2001, Balzano et al. 2002). Though the studies did not mention whether the emergent wetlands were dominated by native vegetation, Brown and Veneman (2001) noted "plant communities in replicated wetlands differed significantly from those in wetlands they were designed to replace" in terms of the number and percent cover of species in general and the number and percent cover of wetland species.

Compensatory mitigation may often result in a change of wetland type. The studies examining Cowardin classes at compensation wetlands found a net increase in open water/aquatic bed habitats. Though the reasons for this change are not clear for all studies, several studies indicated that open water/aquatic bed resulted from an inability to establish the proposed Cowardin class. Another possible reason may be that open water is relatively easy to establish given adequate hydrologic conditions. Furthermore, it is possible that the wetland mitigation design was intended to maximize a limited space by providing a variety of habitat niches, and open water is often considered a key habitat niche for waterfowl. Therefore, it is possible that regulatory decisions may have been biased toward the construction of more open water/aquatic bed/emergent wetland complexes in order to maximize space and achieve an enhancement of functions.

Studies from Washington and Oregon reported a net loss of emergent wetlands. However, many wetlands in the Puget Lowlands of Washington and the Willamette Valley of Oregon that are classified as emergent are wet pastures dominated by non-native grasses. Johnson et al. (2002) did note that 90% of the emergent acreage lost or converted was pasture dominated by non-native species. Converting pastures into other wetland types with a greater diversity of hydroperiod and more structural complexity may therefore represent an opportunity for a net increase in wetland functions over time.

6.8.2.3 Factors Affecting Plants

Several major factors influencing wetland vegetation emerged from the literature:

- Soil and soil disturbance
- Age of the wetland
- Competition and non-native vegetation
- Seed or plant source
- Human manipulation

The studies summarized in this section looked at different parameters in different types of wetlands across the country; therefore, the results are highly variable.

Soil and Soil Disturbance

Five studies indicated that soil conditions at the created or restored wetlands influenced vegetation composition (Brown 1991, Ashworth 1997, Brown and Bedford 1997, Stauffer and Brooks 1997, Brown 1999). Three of these studies discussed the positive effects of adding salvaged or donor hydric soil to created or restored wetlands. Benefits included increased species richness (Brown 1991, Stauffer and Brooks 1997) and significantly higher number and percent cover of wetland species (Brown and Bedford 1997). Stauffer and Brooks (1997) concluded that more organic matter in the hydric soil improved the retention of moisture and nutrients, thereby helping to increase plant cover, density, and species richness. Another study, involving dike removal to restore a site, observed that disturbance of the soil resulted in vegetation dominated by cattails (Brown 1999).

Age of the Wetland

The effect of age on the vegetation of created and restored wetlands was noted in various studies (Reinartz and Warne 1993, Magee et al. 1999, Moore et al. 1999). Created and restored wetlands less than three years old differed, in terms of floristic composition, from sites three years and older (Reinartz and Warne 1993, Magee et al. 1999). Older sites had higher mean total plant cover and mean cover of native wetland species (Reinartz and Warne 1993). Moore et al. (1999) found that age, in addition to sedimentation, resulted in:

- A decrease in open water and water depth
- An increase in emergent and woody cover
- An increase in the number of species
- An increase in wetland species richness

In western Washington, Celedonia (2002) investigated the age at which canopy convergence occurs. The study found that "aerial [woody] cover increases with age until year 8 and remains constant into years 10-11." The author noted, "80% cover is generally achieved by year 8, and perhaps as early as year 7." In addition, the study found that native woody cover was strongly correlated with the density of stems greater than 6.5 feet (2 m) tall, such that percent cover increased as stem density increased up to about 2,100 stems per acre. Sites with densities higher than 2,100 stems per acre generally had greater than 90% woody cover (Celedonia 2002).

Competition and Non-Native Vegetation

The effect of competition on vegetation has been examined in several studies. McLeod et al. (2001) determined that an existing willow canopy did not detrimentally affect the survival of three under-planted tree species in the southeastern United States. Budelsky and Galatowitsch (2000) experimented with hairy sedge (*Carex lacustris*). The authors concluded, "*C. lacustris* can produce dense stands under a primarily annual weed community within two to three growing seasons, but that reed canary grass can preclude successful establishment of *C. lacustris*" (Budelsky and Galatowitsch 2000).

Research has identified two factors that affect competition with non-native species:

- **Shrub density.** Celedonia (2002) observed, "greater shrub layer densities were associated with less reed canarygrass." The author suggests that an initial planting of a very dense shrub layer (e.g., more than 3,000 stems per acre) may help to preclude domination of reed canarygrass (*Phalaris arundinaceae*).
- **Land use.** Magee et al. (1999) found that "the number of introduced and invasive/introduced species per site increases significantly with more intensive land use."

A few studies investigated how many compensation projects experienced problems with invasive species or how many non-native species occurred on sites. In Washington State, Johnson et al. (2002) noted that 61% of compensatory mitigation sites had at least 25% of the site dominated by non-native species. Celedonia (2002) found that nearly half of the sites he visited in Washington had greater than 10% cover of reed canarygrass. In a study conducted by the Michigan Department of Environmental Quality (2000), "8% of mitigation sites were found to have a problem with invasive species" (defined as constituting 10% or more of the vegetation community).

In the Portland metropolitan area of Oregon, a study of vegetation at compensatory mitigation wetlands observed that non-natives composed more than half of the species present and "nine of the 14 most common taxa were invasive introduced species" (Magee et al. 1999).

Seed or Plant Source

The seed or plant source has been identified as important for restored wetlands (Reinartz and Warne 1993, Galatowitsch and van der Valk 1995). Restoration wetlands seeded

with native wetland species had higher diversity and richness and less cover by cattails than the unseeded wetlands (Reinartz and Warne 1993). Emergent perennial species rapidly recolonized restoration wetlands possessing a viable refugium of wetland plant species (e.g., present in existing ditches) that spread through vegetative rooting (Galatowitsch and van der Valk 1995). The importance of proximity to a seed source was mentioned by Reinartz and Warne (1993) but discounted by Galatowitsch and van der Valk (1995).

Human Manipulation

A study conducted in the Willamette Valley of Oregon examined the response of wetland vegetation to three techniques for the restoration of wet prairie: burning, hand removal, and mowing (Clark and Wilson 2001). Results indicated that:

- **Burning** significantly reduced the survival and percent cover of woody species and non-native forbs (e.g., common St. John's-wort [*Hypericum perforatum*]), increased flowering of slender rush (*Juncus tenuis*), and increased cover of native forbs (e.g., Spanish-clover [*Lotus purshiana*] and marsh speedwell [*Veronica scutellata*]), but decreased flowering of tufted hairgrass (*Deschampsia cespitosa*), the dominant wetland prairie grass.
- **Hand removal** significantly reduced cover by woody species and non-native forbs, increased cover of native forbs, but increased flowering of non-native grasses (e.g., velvet grass [*Holcus lanatus*] and sweet vernal grass [*Anthoxanthum odoratum*]).
- **Mowing** had no effect on cover of woody species, but it increased the flowering of non-native grasses and significantly increased flowering of slender rush.

The authors concluded that though "no treatment was clearly superior in fulfilling the restoration objectives" mowing with removal of cut material was specifically not recommended (Clark and Wilson 2001).

6.8.3 Soil Characteristics

Soils are a critical component of wetlands. Soil characteristics can influence the growth and development of vegetation as well as the ability of wetlands to perform certain water quality functions. Researchers have investigated several factors related to wetland soil characteristics at compensatory wetland mitigation sites, including:

- Organic matter content
- Bulk density (compaction)
- Particle size
- Nitrogen content

Several authors used the approach of comparing soil conditions of created wetlands with reference wetlands, while one study compared treatment plots to control plots at created wetlands. None of the articles on soil characteristics involved non-regulatory projects, and all were outside the Pacific Northwest.

Results consistently indicated that the soil of created wetlands had lower organic matter content than reference wetlands (Brown 1991, Bishel-Machung et al. 1996, Streever et al. 1996, Shaffer and Ernst 1999, Whittecar and Daniels 1999, Stolt et al. 2000). This was the case regardless of the Cowardin class or hydrogeomorphic class (Bishel-Machung et al. 1996). However, Shaffer and Ernst (1999) found a negative relationship between inundation and organic matter content of the soil. For example, a wetland covered by standing water for a long duration had less organic matter than a wetland with less inundation. The authors believed this could be due to excavation in these sites, thereby resulting in a loss of the upper soil layers and more water ponding.

In studies examining created wetlands from one to 11 years old and one to eight years old, the age of the created wetlands did not have an effect on organic matter content of the soil (Shaffer and Ernst 1999, Bishel-Machung et al. 1996). Concentrations of organic matter were relatively uniform between surface and subsurface samples. This indicated that accumulation of organic matter was either not occurring or was occurring so slowly it was not detectable (Bishel-Machung et al. 1996, Shaffer and Ernst 1999).

Stauffer and Brooks (1997) examined the effect of adding organic soil amendments to created wetlands. The authors found that plots treated with "salvaged marsh surface" (hydric topsoil) and leaf litter compost contained more organic matter than untreated, control plots. After two growing seasons, soil organic matter remained higher in plots treated with organic soil amendments.

Studies looking at particle size, bulk density, and nitrogen content found that soils in created wetlands had more sand, higher bulk densities (more compacted), and a lower nitrogen content than reference wetlands (Bishel-Machung et al. 1996, Whittecar and Daniels 1999, Stolt et al. 2000). In combination with low organic content, the soil characteristics of created wetlands could hinder plant establishment and growth (Whittecar and Daniels 1999, Stolt et al. 2000), denitrification and pollutant trapping (Stolt et al. 2000), and redox conditions (Bishel-Machung et al. 1996), thereby influencing microbial activity (Whittecar and Daniels 1999).

In contrast, Gilliam et al. (1999) found that redox levels and nitrogen content (in the form of ammonia) at an eight-month-old created wetland were comparable to a reference wetland after the created wetland was inundated. However, pH, phosphorus, manganese, magnesium, and zinc did not change noticeably at the created site. The authors concluded that eight months was "an insufficient period of time for a complete change toward hydromorphic soils."

6.8.4 Water Quality

Most of the water quality studies investigated the ability of created or restored wetlands to retain sediment, phosphorus, nitrogen, or some combination. One study compared water quality attributes at created and reference wetlands (Streever et al. 1996). None of the studies were conducted in the Pacific Northwest.

6.8.4.1 Comparison of Water Quality at Created and Reference Wetlands

Streever et al. (1996) determined that created wetlands had higher pH and conductivity than reference wetlands. The authors hypothesized that the amount of organic matter in the soil is related to pH and conductivity: "because decomposition of organic material releases CO2, lower pH values would be expected in natural systems with well-developed organic soils. A well-developed organic substrate may isolate surface water from underlying sand and rock, leading to decreased dissolution of minerals and lower conductivity." (See the previous discussion of soil characteristics in Section 6.8.3.)

6.8.4.2 Sediment Removal

Findings related to retention of sediment by created wetlands include the following:

- Wetlands created adjacent to roads were effective at retaining sediment, such that inflow culverts were clogged by accumulated sediment at a couple of sites (Moore et al. 1999).
- Mitsch (1992) found that a created wetland retained 90% of sediments, while a reference wetland retained 3%. The actual amount of sediment retained depends upon the loading rate.
- Fennessey et al. (1994) investigated the location within a created wetland where sediment was retained. Rates of sediment deposition, in general, were highest near the inflow and decreased as distance from the inflow increased, "except when outflow ceased, in which case the maximum sedimentation often occurred near the outfall." Open water areas also had higher sediment deposition than vegetated areas, which restricted flow. The authors observed that vegetation seems to present a barrier to water and sediment flow and, therefore, the study "did not illustrate the conventional belief that the presence of vegetation enhances sedimentation." The authors concluded, "deeper open water areas are more conducive to sediment accumulation than are shallower open water areas that are more easily subjected to wind-driven and biological sediment disturbances and subsequent re-suspension."

6.8.4.3 Nutrient Removal

In several studies, phosphorus retention at created or restored wetlands ranged from 16 to 96% (Mitsch 1992, Mitsch et al. 1995, Niswander and Mitsch 1995, White et al. 2000). In all but one of these studies, created/restored wetlands retained at least 53% of phosphorus (Mitsch 1992, Mitsch et al. 1995, White et al. 2000). The percent of retention varied depending on:

- Whether the wetland experienced high or low flows (Mitsch 1992, Mitsch et al. 1995)
- The configuration of the outflow
- The amount of time water was retained in the wetland (Niswander and Mitsch 1995)

White et al. (2000) mentioned that a restored wetland's capacity for phosphorus retention is limited. Sediments near the wetland inflow had a limited ability for additional uptake of phosphorus. However, approximately 66% of the marsh sediments still had a high capacity for uptake. The authors concluded, "future treatment efficacy may decrease if the remaining sediments become saturated. Continued high P [phosphorus] loading to the marsh may lead to eutrophication problems and downstream P export from the wetland."

Romero et al. (1999) found that total nitrogen retention was 30 to 91% at four restored wetlands. The authors attributed this to the high retention of dissolved inorganic nitrogen, while the retention efficiencies for particulate and dissolved organic nitrogen were much lower. The authors observed no significant difference between nitrogen retention and the age of the restored wetland.

Woltemade (2000) examined the factors that affect the ability of a created or restored wetland to retain nutrients. The most critical design elements for wetlands constructed to treat agricultural runoff were determined to be the retention time (amount of time that water is retained in the wetland) and the wetland-to-watershed ratio (size of the wetland compared to the size of its contributing basin):

> *If nutrient and sediment concentrations are to be reduced to acceptable levels on a landscape scale, drainage water must be retained for at least one to two weeks within wetlands before being discharged into streams. Monitoring of restored wetlands indicates that the longer the retention time, the greater the water quality benefits . . . Ultimately, the appropriate size of a restored wetland will depend on the contaminant of greatest local concern that requires the longest retention time for its degradation, and on the percent reduction of this contaminant that is required seasonally, annually, or interannually (van der Valk and Jolly 1992).* (Woltemade 2000).

6.8.5 Water Quantity

No studies were found that discussed the ability of created or restored wetlands to perform water quantity functions, such as decreasing downstream erosion or reducing peak flows, or that mentioned factors influencing a wetland's ability to perform water quantity functions.

Two studies compared the water regime of compensatory mitigation wetlands with reference wetlands. Both found that the compensatory wetlands had more standing water for a longer period (Shaffer et al. 1999, Cole and Brooks 2000).

6.8.5.1 Using Hydrogeomorphic (HGM) Classification to Study Water Regime at Mitigation Sites

Differences in the water regime between existing wetlands and mitigation wetlands have been examined by several researchers in the Pacific Northwest and elsewhere. The researchers used the hydrogeomorphic (HGM) classification to compare the water regimes of existing wetlands with those of mitigation wetlands. As described in Chapter 2, the HGM classification is based on the position of the wetland in the landscape (geomorphic setting), the wetland's water source, and the flow and fluctuation of the water once in the wetland. These are some of the major environmental factors that control wetland functions (National Research Council 1995).

Gwin et al. (1999) focused on HGM classifications of wetlands to determine how compensatory mitigation was affecting the wetland resource in and around Portland, Oregon. Classification of reference wetlands resulted in three regional HGM classes: slope, riverine, and depressional. However, classification of mitigation wetlands

> *required development of new, atypical HGM classes to describe the unique combinations of site morphology and landscape setting found in these wetlands:*
>
> - *depression-in-riverine setting,*
> - *in-stream-depression, and*
> - *depression-in-slope setting* (Gwin et al. 1999).

Gwin et al. (1999) characterized atypical classes by:

- Exaggerated depressional morphology with steep banks
- Large areas of open and/or deep water
- A large berm isolating the wetland from an adjacent stream channel
- Excavation within the stream channel producing an open water area wider and deeper than the original stream

In Washington 35% of compensatory mitigation projects resulted in wetlands of an atypical HGM class (Johnson et al. 2002). In Portland, Gwin et al. (1999) found that almost all of the enhanced wetlands and nearly half of the created wetlands resulted in an atypical HGM class.

What are the hydrologic consequences of creating atypical wetlands in the landscape? Shaffer et al. (1999) examined hydrologic conditions in reference and mitigation wetlands in the Portland metropolitan area. The study compared the regional HGM classes identified by Gwin et al. (1999)—slope, riverine, and depressional—with the atypical classes for mitigation wetlands—depression-in-riverine setting, depression-in-slope setting, and in-stream-depression. The results indicated significant differences. For example, slope wetlands had the lowest extent, depth, and duration of inundation, "while depression-in-slope wetlands had the highest water levels and greatest extent/duration of inundation" (Shaffer et al. 1999).

Similarly, Cole and Brooks (2000) noted that created wetlands were dominated by open water, while "most naturally occurring mainstem floodplain wetlands in central Pennsylvania are vegetated with very little open water." The authors concluded, "in the rush to make sure there is some water in mitigation wetlands we have gone too far in keeping sites inundated. In reality, many wetlands are merely saturated, or much drier" (Cole and Brooks 2000).

Schaffer et al. (1999) state:

> *Unless wetlands are restored or created in a manner that reproduces the hydrogeomorphic characteristics of naturally occurring wetlands in a region, management activities are unlikely to maintain or replace hydrologic and other valued functions of wetlands*.

Similarly, Cole and Brooks (2000) conclude:

> *The ecological consequences of a different hydrologic regime are clear. Standing water will promote anaerobic conditions in the soil, and the resulting soil chemistry will be defined by anaerobic pathways (Mitsch and Gosselink 1993). When combined with other common construction effects (e.g. soil compaction), this leads to difficult conditions for plant community establishment.*

In addition, water regimes exhibiting extensive areas of open water in mitigation wetlands hindered the formation of soil organic matter (Shaffer and Ernst 1999).

6.8.6 Summary of Key Points

- Functions performed and characteristics produced by created and restored wetlands differed from those performed and produced by reference wetlands, except water quality functions, which appeared to be performed in a similar capacity.

- Most studies determined that reference wetlands provided habitat for a greater diversity or abundance of wildlife than created or restored wetlands. Birds were an exception since half of the studies found no difference between created/restored sites and reference wetlands, particularly for ducks.

- A variety of factors appeared to influence the abundance and diversity of wildlife at created or restored wetlands: development of vegetation communities, particularly emergent vegetation communities; age of the wetlands, which is often associated with the development of vegetation communities; and availability of a food source, often invertebrates, which is also often associated with the development of vegetation communities.

- Amphibian communities were affected by additional factors, such as the hydroperiod of the wetland, the presence of fish, distance to other wetlands, connectivity between terrestrial and wetland habitats, and surrounding land uses.

- Created and restored wetlands have different vegetation characteristics and plant communities than reference wetlands. A few studies found that certain plant communities, such as sedge meadows, may require many years to develop if at all.

- Compensatory mitigation is producing more acreage of open water wetlands than was lost. The ability of compensatory mitigation to produce other Cowardin classes varied.

- Several major factors were found to affect vegetation and plant communities, including the age of the wetland (older created/restored sites had higher percent cover of emergent and woody species than younger sites); soil conditions (positive effects on vegetation resulted from adding hydric topsoil); competition (reed canarygrass can be problematic when attempting to establish emergent vegetation); and a source of native seeds or plants (this can speed up recolonization and increase diversity).

- Created, restored, and enhanced wetlands had less organic matter than reference wetlands. This could be due to excavation of surface soil layers during project installation. In addition, organic matter at compensation wetlands did not appear to accumulate over time. Plant establishment at compensation sites could be hindered by the low organic content in conjunction with soils that were found to be sandier, more compacted, and lower in nitrogen.

- Created and restored wetlands were comparable to reference wetlands at retaining sediments, phosphorus, and nitrogen. Factors affecting sediment and nutrient retention included the volume of water flowing into the wetland, the length of time water remains in the wetland, and the size of the wetland compared to the size of the basin.

- Some compensatory mitigation wetlands produced significantly different HGM classes than were present in reference wetlands. This has resulted in wetlands that have more inundation for a longer duration than reference wetlands.

6.9 Reproducibility of Particular Wetland Types

This section discusses findings from the literature regarding the ability to restore, create, or enhance certain wetland types, such as bogs and fens, vernal pools, alkali wetlands, and mature forested wetlands.

6.9.1 Bogs and Fens

Bogs and fens are characterized by their highly organic soil conditions, water regimes, and water chemistries. Studies of bog and fen restoration in Northern Europe and Canada concluded that restoration may not be possible due to "irreversible changes of the biotic and abiotic properties" (Schouwenaars 1995, Schrautzer et al. 1996). This includes soil compaction and eutrophication (Grootjans and van Diggelen 1995, Schrautzer et al. 1996, Wind-Mulder and Vitt 2000) and other alterations to bogs resulting from drainage, peat harvesting, pollution, and agricultural practices (National Research Council 2001).

The studies mentioned difficulties in restoring bog vegetation communities (Bolscher 1995, Grosvernier et al. 1995, Schouwenaars 1995, Schrautzer et al. 1996), water regime (Grootjans and van Diggelen 1995, Schouwenaars 1995), and/or water chemistry (Wind-Mulder and Vitt 2000). Major conclusions include:

- Restore the water regime and the vegetation community will follow (Grootjans and van Diggelen 1995, Grosvernier et al. 1995).
- Prior to any restoration activity, the chemical state of the bog must be assessed. This influences the vegetation community and will, therefore, dictate the development of a restoration plan (Wind-Mulder and Vitt 2000).
- "Hydrological research may be crucial for a correct assessment of perspectives for rewetting" (Schouwenaars 1995). Prior to restoration it is necessary to determine the reason for a low water table because this affects the activities that will be required to restore a suitable water regime for the desired vegetation communities (Schouwenaars 1995).
- Bogs that were restored by rewetting and tree removal "differed from those of natural raised bogs, particularly in having a taller and denser vegetation, a smaller range of moisture gradient and a more uniform vegetation physiognomy. Rewetted bogs did not have an undulating surface relief of hummocks and hollows" (Bolscher 1995).
- The best chance for restoration lies with restoring the least disturbed or damaged bogs or fens (Grootjans and van Diggelen 1995, Schrautzer et al. 1996).

- Restoration of bogs or fens will not yield rapid results (Grootjans and van Diggelen 1995).
- "Research has demonstrated that natural recovery of the moss surface following harvesting takes about 20 years (Elling and Knighton 1984)" (National Research Council 2001).

In terms of creation, research indicates that in reference systems organic soil (peat) accumulates at 0.1 to 3.8 mm per year (National Research Council 2001). At this rate it would take from 7 to 250 years for just 1 inch of peat to accumulate.

No information was available on the success or compliance of bogs or fens that were restored or created as wetland compensation. However, the literature suggests that bogs and fens cannot be reproduced within a regulatory timeframe.

6.9.2 Vernal Pools

Vernal pools are characterized by their short duration of inundation (National Research Council 2001). Thus, in order to reproduce a vernal pool, a site with a suitable substrate must be found and the correct depth and hydroperiod must be created or restored (National Research Council 2001). "In a long-term study of California vernal pools that were created by excavating depressions near natural pools, the hydroperiods did not converge with those of the reference systems until year 10 (Zedler et al. 1993)" (National Research Council 2001). If the hydroperiod is too long, the result will be an emergent marsh or an open water or aquatic bed system. If the site has inadequate substrate or is too shallow, the result may be upland with no inundation.

In terms of compliance, De Weese (1998) examined over 1,500 created vernal pools in California. She found that 83% of projects were in permit compliance, 96% met their hydrologic performance standards for depth of inundation, and 69% met vegetation performance standards. Seventy-two percent of projects were compared with reference vernal pools to determine their biological viability, while 35% of projects required some site remediation.

Guidance on construction has helped to transform the steep-sided "bathtubs" into pools that more closely mimic reference pools with gradual, vegetated slopes (De Weese 1998). De Weese (1998) concluded, "The art and science of constructing vernal pools have greatly improved over the past eight years [1987 to 1994]."

The literature suggests that, in California, vernal pools may be reproduced under the right conditions. However, the right conditions typically occur where vernal pools already exist, so creation of new pools merely increases the density of pools in an area (National Research Council 2001).

No information was found on the reproducibility of vernal pools in Washington.

6.9.3 Alkali Wetlands

No information was found that addressed the reproducibility of alkali wetlands.

6.9.4 Mature Forested Wetlands

Though studies have found that forested wetlands can be reproduced in Washington (Celedonia 2002, Johnson et al. 2002), **mature** forested wetlands have not been successfully reproduced simply because of the time necessary for the trees and the structural characteristics of the forest to mature (National Research Council 2001). Enhanced and created sites that have been planted often have a high density of stems to rapidly provide woody cover and shade out invasive species in the understory (Celedonia 2002, National Research Council 2001). Unless these sites are thinned, they will not reproduce the attributes of mature forested reference wetlands (National Research Council 2001).

6.9.5 Summary of Key Points

- The reproducibility of some wetland types is generally dependent upon time. For example, bogs, fens, and mature forested wetlands require several decades, at a minimum, to develop the structural, chemical, biological, and hydrological attributes that characterize these wetland types.
- Studies suggest that vernal pools, at least in California, may be reproducible under the right conditions.

6.10 Suggestions from the Literature for Improving Compensatory Mitigation

A number of reports and articles suggested or recommended changes that could be made to help improve the effectiveness of compensatory wetland mitigation or alleviate problems that were frequently encountered. The recommendations described below are those of the authors of the literature sources cited, not the agencies or staff who have synthesized the information in this volume.

The scientific literature contained recommendations that fall into three main categories:

- Recommendations for regulators of compensatory mitigation, including guidance on mitigation plans and monitoring reports, compliance tracking and enforcement, and alternative mitigation options
- Recommendations for site selection and design, including comprehensive wetland planning, baseline monitoring, hydrologic analysis, and considerations for site design

- Recommendations for implementing compensatory mitigation, including having a wetland biologist on-site to oversee construction activities, performing monitoring and maintenance of the site

The scientific literature provided more extensive information on additional topics:

- Performance standards
- Compensatory mitigation using a watershed approach
- Mitigation banking and in-lieu fees

Each of these is discussed below.

6.10.1 Regulatory Improvements

Of the suggestions provided by the scientific literature, the majority focused on elements that regulatory agencies should address (Table 6-11), such as:

- Improving guidance for every step of the mitigation process, from avoidance and minimization to submitting a monitoring report for a compensation wetland. This should help regulators with decision-making and provide applicants and consultants with more predictability
- Adjusting replacement ratios to reflect the risk of failure
- Requiring financial assurances or performance bonding
- Protecting all compensatory mitigation sites in perpetuity with a legal mechanism, such as a deed restriction or conservation easement
- Increasing regulatory follow-up and enforcement of compensatory mitigation projects, including developing and maintaining a database and filing system, allocating staff to perform compliance and enforcement activities, and implementing reviews of regulatory program performance
- Developing and implementing alternative mitigation options, such as advance mitigation, mitigation banking, and in-lieu fees

Table 6-11. Suggestions from the literature for regulatory improvement.

Suggestion	Reference No. [a]
Improve mitigation sequencing (i.e., avoidance and minimization)	5, 19, 29, 2, 10
Improve guidance for compensation projects, focusing on replacing functions as well as area	22, 29, 10
Improve site selection criteria. Site selection should be based on a watershed scale to maintain diversity, connectivity, and a balance of upland and wetland	5, 12, 15, 29, 2, 10
Improve goals, objectives, and performance standards, so that they are measurable, meaningful, and enforceable	13, 5, 31, 20, 19, 17, 9, 29, 10
Standardize report format and elements for mitigation plans and monitoring reports, including an implementation schedule	13, 7, 8, 20, 17, 15, 2, 10
Adjust (increase) replacement ratios to reflect the risk of failure. This should be based on the level of success of previous projects	1, 2, 18
Require performance bonding/financial assurances	13, 5, 8, 20, 15, 29, 2, 18
Require that compensation wetlands be protected in perpetuity with some kind of legal mechanism, such as a deed restriction or conservation easement	20, 29
Improve regulatory follow-up and enforcement of compensatory mitigation projects	5, 20, 19, 1, 30, 17, 15, 9, 29, 10, 18
Develop and maintain a permit/compensatory mitigation project tracking database and filing system	5, 7, 20, 19, 1, 17, 9, 15, 29, 2
Allocate staff for compliance and enforcement	5, 19, 17
Implement regular reviews of regulatory program performance	7, 19, 2
Implement studies of cumulative wetland loss (beyond what is recorded for regulatory permitting programs)	19, 1
Develop and implement alternative compensatory mitigation options: in-lieu fees, mitigation banking	20, 19, 1, 15, 29
Perform the compensatory mitigation in advance of the wetland loss	30, 29, 10, 18

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

6.10.2 Improving Site Selection and Design

The scientific literature also suggested site selection and design considerations (Table 6-12), including:

- Using a watershed approach to improve site selection
- Prioritizing wetland restoration
- Performing baseline monitoring of the wetland to be lost, identifying the wetland types and functions so that they can be replaced more effectively

- Performing baseline monitoring of the areas proposed for compensation to document the existing conditions and level of function
- Performing a hydrologic analysis for compensation wetlands to identify where the water will come from, how it will get to the site, and what the extent and duration of inundation or saturation will be
- Designing the compensation site to be self-sustaining and incorporating or simulating natural processes and structures, such as hydroperiods, slopes, shorelines, soils, topography, and vegetation

Table 6-12. Suggestions from the literature for improving site selection and design.

Suggestions	Reference No. [a]
Ensure that compensation wetlands will have a suitable source of water and compatible adjacent land uses	5, 28, 17, 29, 2, 10
Use a watershed approach to select compensation sites and support comprehensive wetland planning	19, 1, 28, 17, 29, 10
Prioritize restoration as the first choice for compensatory mitigation	17, 15, 29
Design compensatory mitigation wetlands to be self-sustaining and incorporate natural processes whenever possible	33, 29
Perform baseline monitoring of wetlands to be lost and areas proposed for compensatory wetland mitigation. Monitoring should characterize hydroperiod, soils, water quality, macroinvertebrates, and wetland functions	13, 5, 31, 20, 29, 10
Perform hydrologic analysis: identify hydrologic source, how water will get to the site, the intended depth and duration of inundation, and demonstrate that water source will be reliable and adequate Determine appropriate hydroperiod/hydrologic inputs early in the design stage, so that the water levels of the compensation wetland dictate how to design the building sites and roads, rather than letting the upland development create poor wetland conditions (too wet or too dry)	5, 31, 16, 28, 2
Grade slopes to be as gentle as possible; they should match the slopes of adjacent natural wetlands	6
Provide heterogeneous topography. For example, simulate microtopographic "mound and pool" features (e.g., wind-thrown or toppled trees)	23, 29
Incorporate native upland ecosystems into compensatory mitigation sites	5, 29
Deconsolidate soils to reduce compaction and amend to insure adequate soil organic matter (e.g., 2 inches of coarse sand and 4 inches organic compost, natural hydric muck, or topsoil)	6, 16, 28, 32
Minimize human encroachment by planting dense vegetation around the site or installing fences	26, 20, 16
Establish rapid canopy convergence and limit invasive species infestations by planting trees and shrubs at specific densities	25
Demarcate the site with signs and boundary markers	20, 16

[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.

6.10.3 Improving Implementation

Compensatory wetland mitigation projects would be greatly improved if they were implemented as designed (Johnson et al. 2000, Balzano et al. 2002). The scientific literature provides numerous suggestions for improving implementation (Table 6-13), such as:

- Having a wetland biologist on-site during construction
- Monitoring the compensation wetland
- Maintaining the compensation wetland to avoid problems, or identify and manage them early in the development of the site

Table 6-13. Suggestions from the literature for improving implementation.

Suggestion	Reference No. [a]
Wetland biologist on-site to oversee construction or train/educate contractors and to authorize and document any necessary changes	5, 31, 9, 2
Monitoring of mitigation sites should characterize baseline, construction, as-built, and post-construction conditions. Monitoring reports should include a section on lessons learned	13, 6, 5, 24, 7, 31, 8, 20, 19, 15, 9, 2, 10
Monitoring parameters and methods should be specific to a project's goals, objectives, and should include: project size, shape, topography, hydroperiod, water quality, flora, and fauna	5, 17, 33
Duration of monitoring depends on the size of compensation wetland, the proposed wetland type (e.g., Cowardin class), and the likelihood of success. Anywhere from 3 to more than 15 years were suggested	5, 34, 35, 17, 15, 10
Monitor hydrology during the first growing season to characterize the site's hydroperiod. Develop and implement a planting plan after the hydroperiod has been characterized	27
Perform long-term monitoring after a project has been deemed successful to keep track of it over time, study how it matures, use it as model for other sites	5, 28
Maintenance of compensatory mitigation sites, including a contingency plan for how to address problems. Maintenance should focus on controlling invasive species, providing irrigation, replacing dead plants, correcting slopes and topography	13, 24, 31, 8, 2, 10
[a] See Table 6-1 for a listing of literature sources that correspond to each reference number.	

6.10.4 Performance Standards

Performance standards, performance criteria, success criteria, success measures, standards of success, and other terms all refer to regulatory conditions used to determine how effective a mitigation project is at meeting regulatory requirements, which may or may not include compensating for wetland loss. Ideally performance standards should serve as "measurable benchmarks used to evaluate the development of ecological characteristics associated with specific wetland functions" (Azous et al. 1998). Performance standards allow regulators to determine if a compensatory mitigation project

has fulfilled its goals, and also provide a mechanism for regulators to implement enforcement actions against unsuccessful projects (Streever 1999).

As explained in Chapter 2, wetlands differ in how they function by geomorphology and water regime and other characteristics. Compensatory wetland mitigation projects, likewise, exhibit considerable variability with different types of wetland compensation (creation, restoration, etc.). The variability makes it difficult to develop and require universal performance standards, yet in the absence of some kind of uniformity, performance standards that are approved can lack meaning.

6.10.4.1 Shortcomings of Existing Performance Standards

Sheldon and Dole (1992) performed a study of eight compensatory mitigation projects in King and Snohomish Counties in Washington. The authors observed that "none of the goal statements provided a quantifiable method of determining success, thus they provided no means for an agency to assess success/failure or to require remediation." The Michigan Department of Environmental Quality (2000) similarly found, "The practice of including no specific performance standards, or only very general performance standards (regarding the size and possibly the type of wetland to be constructed), resulted in many unenforceable permits and contributed to the poor quality mitigation wetlands."

Johnson et al. (2000), in their study of 45 compensatory mitigation wetlands, noted some problems with performance standards, such as:

- Standards that are too general or "easy to attain" and, therefore, are not indicative of ecological development at a site
- Standards that are not measurable and, therefore, cannot be used to evaluate the success or compliance of projects
- Standards that contain confusing or ambiguous language and, therefore, result in inaccurate assessment or preclude assessment

Approved mitigation projects can also lack performance standards for important wetland functions or conditions. Breaux and Serefiddin (1999) discovered in their review of 110 projects in San Francisco, California, that only 22% had quantitative standards focusing on hydrological parameters. Johnson et al. (2000) reviewed 179 performance standards from 36 projects and observed that 8% of the performance standards related to hydrological conditions. The Michigan Department of Environmental Quality (2000) found that "none of the permits examined contained any specific criteria regarding vegetation or hydrology by which the mitigation wetland could be judged for success or failure." Johnson et al. (2002) noted that most of the projects evaluated in their study of 24 compensation wetlands lacked basic standards for wetland area, water regime, area of Cowardin classes, percent cover of native wetland vegetation, and maximum percent cover of invasive vegetation.

Breaux and Serefiddin (1999) argue, "In seasonal wetlands, hydrology clearly ought to be the reigning criterion given that the successive presence and absence of water is the defining characteristic of a seasonal wetland." However, the authors go on to admit, "there is no agreement as to what the specific hydrological criterion should be."

6.10.4.2 Use of Reference Wetlands in Developing Performance Standards

Brinson and Rheinhardt (1996) state that "the proper use of reference wetlands removes potential bias and provides the foundation for more objective functional-assessment procedures…reference wetlands should be central to the development of standards against which impacts to wetlands and restoration efforts are evaluated."

Azous et al. (1998) also support the use of reference wetlands:

> *By collecting data on the ecological characteristics associated with reference wetlands, and created or restored wetlands, standards of comparison can be established by which to judge the development of wetland characteristics in compensatory mitigation projects. The use of regional reference wetland characteristics provide greater assurance that project performance standards will be reasonable (i.e.: attainable) and useful gauges of the development of wetland functions.*

For example, a compensation wetland might have a goal to provide amphibian habitat by the end of the monitoring period. Based on an evaluation of 24 depressional, flow-through, reference wetlands in the Puget Lowlands of western Washington, Azous et al. (1998) proposed performance standards to determine if amphibian habitat had successfully been established. "The standards include specific guidelines for planning and designing mitigation projects to provide preference for the establishment of amphibian breeding, feeding, and refuge habitats." A few performance standards suggested by the authors include:

- "Wetlands created for amphibian habitat should have thin-stemmed emergent plants comprise at least 30% or more of the total wetland area."
- "Limit mean water level fluctuation (WLF) to 21 cm annually" (Azous et al. 1998).

However, Whittecar and Daniels (1999) mention a problem with using reference wetlands to develop benchmarks or performance standards for compensatory mitigation:

> *[U]nlike the mitigation site, reference wetlands coexist with landforms that may have required thousands of years to form (Brinson et al. 1995). Each wetland has a history that influences modern functions. Many of these functions will not redevelop in the new wetland within a time span acceptable to regulatory constraints without thoughtful planning and careful attention to construction.*

Ehrenfeld (2000) recommends that reference sites be identified in urban areas and used to develop attainable performance standards for compensatory wetland mitigation projects that are also located in urban areas. The author states: "Measures of restoration success and functional performance must start with an appreciation and assessment of the particular conditions imposed by the urban environment."

6.10.4.3 Longer Period Needed to Evaluate Projects

Part of the problem with developing achievable performance standards is that monitoring periods or regulatory timeframes for the majority of compensatory mitigation projects are relatively short (five to 20 years). The "success" or compliance of compensatory mitigation projects is, therefore, determined or evaluated when the site is still relatively young and immature (Kentula 1995, Mitsch and Wilson 1996). If projects are to be evaluated within five to 20 years, then they should be compared to other compensatory mitigation projects. Kentula (1995) suggests comparing "wetland creation and restoration projects to each other and to similar, naturally occurring wetlands to define standards for project performance over time." She describes an approach for developing performance standards based on monitoring information from previous projects. "In this way, we can be assured that new projects are doing at least as well as past projects."

Celedonia (2002) implemented Kentula's approach by conducting a study of 29 compensatory mitigation projects from six to 11 years old in the lowland wetlands of western Washington. Time series curves were created from the data to determine at what point in time projects could be expected to meet certain vegetative standards, such as percent areal cover of woody vegetation. Based on the data, the author proposed that by year eight a mitigation site could attain 80% cover of native woody vegetation.

6.10.5 Compensatory Mitigation Using a Watershed/Landscape Approach

In the context of compensatory mitigation, a watershed approach means:

> *to recognize that management of wetland types, functions, and locations requires structured consideration of watershed needs and how wetland types and location serve these needs. A watershed approach means that mitigation decisions are made with a regional perspective, involve multiple agencies, citizens, scientists, and nonprofit organizations, and draw upon multiple funding sources (e.g., permittee-responsible, mitigation banks, and in-lieu fees). A watershed approach means that permitting decisions are integrated with other regulatory programs (e.g., storm water management or habitat conservation) and nonregulatory programs (e.g., conservation easement programs)* (National Research Council 2001).

Bedford (1996) explained the need for a watershed/landscape approach as follows:

> *From a policy perspective, the central issue in wetland mitigation is not the effects on a single site but the cumulative effect of numerous mitigation decisions on landscapes. Mitigation must be recognized as a policy that has the potential to re-configure the kinds and spatial distribution of wetland ecosystems over large geographic areas. ... The net effect is the loss of wetland diversity in terms of both hydrologic functions and biological communities, and a consequent homogenization of wetland landscapes. One way to avoid such cumulative effects is to make decisions about individual projects within a framework focused at larger scales (Lee and Gosselink 1988).*

This section describes recommendations from the literature for methods to implement a landscape or watershed-scale approach in order to improve the success of mitigation projects. Further discussion of restoration using a landscape approach is included in Chapter 7 in the context of addressing cumulative impacts to wetlands.

6.10.5.1 Methods for Implementing a Landscape Approach

Three types of watershed planning are described in a report by the National Research Council (2001):

- **Management-oriented** wetland planning, which would replace case-by-case permitting. Decisions about permitting, mitigation sequencing, and the acreage, type, and location of compensation would be made in advance using a watershed approach. This type of watershed plan would require regulatory and non-regulatory programs to be coordinated.

- **Protection-oriented** wetland planning, which is focused on avoiding wetland loss and alteration by identifying wetlands and their ecological value. This type of watershed plan would be used during the mitigation sequencing process.

- **Compensation** wetland planning, which "identifies watershed needs for types, functions, and general locations of wetlands in the landscape in order to establish restoration priorities for both regulatory and nonregulatory programs. …This type of planning might link projects undertaken through both regulatory and nonregulatory programs to secure some desired mosaic of wetlands in the landscape."

Hashisaki (1996) discusses the utility of a landscape-level analysis to examine conditions not just at an impact, compensation, or reference site, but also in the surrounding landscape. A landscape-level analysis "considers the effect of historic, current, and proposed land management practices on the individual functional indicators. . . . In addition to identifying constraints on land management practices, it can be useful in identifying critical preservation and restoration opportunities. Understanding the control

that human activities exert on the disturbance regimes of an ecosystem allows projections about expected future conditions."

Bedford (1996) recommends developing wetland profiles/templates based on the diversity of wetland types that exist in a region as a result of the unique interaction of hydrogeology and climate. By understanding the current and historic wetland types and their relative abundances in a region, decisions regarding compensatory mitigation can be made to help maintain the diversity and hydrologic equivalence.

In some cases, using a watershed approach may result in a watershed plan that identifies all the wetlands in an area and assesses the functions that they perform. Hruby and Scuderi (1995) used this approach for a watershed near Seattle, Washington, that was experiencing development pressure. The goal of the plan was "to ensure that the performance of wetland functions and their societal values continue to be equal to or greater than those currently existing…" (Hruby and Scuderi 1995). Wetland areas targeted for restoration or enhancement were assessed to quantify how much wetland function could be gained. The proposed/potential gain in function through restoration/enhancement could then be used to determine how much wetland function could be lost to development activities in the watershed.

A report by the National Research Council (2001) proposed that "Functional tradeoffs might be considered in the context of the needs of the watershed." A watershed plan would be developed for an area, such that the functions of wetlands proposed for loss or alteration are understood, as well as the needs of the watershed for wetland functions. Functions that are abundant or a low priority in a watershed could be lost and replaced by other functions that are limited or a higher priority in the watershed.

Race and Fonseca (1996) point out that on a national level, a landscape approach to land use and compensation would require the cooperation/participation of thousands or millions of private landowners:

> *Taking a large-scale, ecosystem approach to wetlands management is a significant change in natural resource management policies, one representing a major paradigm shift that will require radical revision in values, management practices, and institutional structures in order to succeed (Cortner and Moote 1994). …Thus, integrating ecologically relevant concepts such as landscape-scale decision criteria need more than good science; it will also require conscious redesign of the entire permitting infrastructure to avoid legal challenges.*

6.10.6 Mitigation Banking

Compensatory mitigation banking and other third-party compensation approaches (in-lieu fee, market-based mitigation) are believed by some to provide part of a solution and have offered new hope for successful compensation wetlands.

Currently, even when wetlands have been avoided or established as compensation they often "have diminished ecological functions from polluted runoff, from changes in hydrologic regimes, and from the fragmentation of the landscape which isolates the wetlands from the surrounding uplands, water, and biological resources of the watershed" (Shabman et al. 1993).

In addition, some federal, state, and local permits for wetland loss do not require compensatory mitigation because the individual impact is so small that compensation is considered impractical, despite the fact that cumulative losses are occurring (Shabman et al. 1993). Finally, even when compensatory mitigation is required there is no guarantee that it will be implemented or successful.

Shabman et al. (1993) outlined a market solution to improve compensatory wetland mitigation. Market-based mitigation approaches start with an entrepreneurial restoration firm seeking to make a profit from selling a product—a wetland ecosystem. If the product is not of a particular quality then it will not sell. For example, if the wetland bank is not in compliance, not meeting its performance standards, or not providing the proposed functions then the regulatory agencies will not accept credits from the bank as compensation for wetland losses. The permit applicant, therefore, will not purchase the "product" of the wetland bank. This is the incentive for the restoration firm to establish a functioning wetland ecosystem.

In addition, a restoration firm can take the time to find a suitable location for the wetland ecosystem that will minimize problems with fragmentation and isolation. Wetland banks can also secure large sites for restoration that would not be feasible on a small project scale.

Once the wetland ecosystem is established, credits or tradable portions of the wetland ecosystem can be made available for purchase to compensate for wetland losses, even wetland losses that were previously too small to require compensation. The availability of bank credits for compensation can also provide efficient permitting since the applicant would not have to worry about getting a mitigation plan approved, and regulators could more readily assess the effectiveness of the compensation.

Mitigation banking in Washington State has been more thoroughly discussed in the *Programmatic Environmental Impact Statement: Washington State's Draft Rule on Wetland Mitigation Banking* (Driscoll and Granger 2001). For additional information on mitigation banking and in-lieu fee programs refer to *Banks and Fees: The Status of Off-Site Wetland Mitigation in the United States* (Environmental Law Institute 2002).

6.10.7 Summary of Key Points

- The scientific literature provided suggestions for improving virtually every aspect of the mitigation process from regulatory guidance and policies to specifications for controlling invasive vegetation.

- Suggestions included measurable, meaningful, and enforceable performance standards; better sites that provide increased benefits due to their location within a watershed; better monitoring of compensatory mitigation wetlands; and measures to increase regulatory follow-up of compensation projects.

6.11 Chapter Summary and Conclusions

Wetland compensatory mitigation has been studied in Washington and elsewhere in the United States for the past 15 years. Considerable data are available to evaluate the effectiveness of compensatory mitigation.

The majority of compensatory wetland mitigation projects described in the literature are neither fully successful nor complete failures, but somewhere in between. While most compensatory mitigation projects were installed, compliance of the projects with permit requirements was generally low due to shortfalls of wetland acreage, failure to achieve performance standards, and a lack of monitoring and maintenance. The few studies that examined the effect of regulatory follow-up suggested that it had a positive influence on the level of compliance and success for compensatory wetland mitigation projects.

There is a general lack of information about the relative effectiveness of the various types of compensation (e.g., restoration, creation, enhancement, etc.). Creation is generally the most frequently used type of compensation, but studies of its effectiveness produced mixed results. Enhancement of wetlands was also frequently used, but few studies examined its effectiveness. Limited studies from Washington indicated a low level of success among enhanced wetlands, primarily due to a minimal gain in functions. However, it may simply take longer for a gain in functions to appear (15 to 20 years rather than five to 10 years). Restoring wetlands was noted as a high priority, but as a type of compensation it is not frequently used. This could be due to the fact that restoration is most cost effective for large projects while most compensation projects tend to be relatively small.

Preservation and a mixture of compensation types appear to be used occasionally. Studies provided limited information on the effectiveness of these types. Two studies from Washington indicated that mixed compensation projects had a higher level of compliance than creation or enhancement, and all mixed projects were moderately successful. The lack of data regarding the effectiveness of preservation is problematic since one of the only studies to look at its effectiveness determined that one large site was predominantly upland habitat. On the other hand, if a site can be confirmed as wetland, or if a mosaic of wetland and upland is determined to be acceptable, preservation of existing wetlands offers no risk of failure and no temporal loss of wetland functions, which are inherent in the other types of compensation. Preservation does, however, result in a net loss of wetland area and possibly functions.

Replacement ratios equalize the trade-off between the wetland being lost and the wetland being provided as compensation by accounting for the risk of failure and temporal loss of functions. Required replacement ratios vary from one state to another, based on the type

of compensation proposed, and based on project-specific circumstances. Replacement ratios actually achieved through compensation were less than what was required, which is to be expected since the ratios are meant to encompass a certain level of failure. However, in some cases this resulted in less than 1:1 acreage replacement.

While on paper, studies indicated that permitting programs have improved over time in terms of wetland acreage required for compensation, studies in the field indicated that compensatory wetland mitigation has resulted in a loss of wetland acreage.

Functions performed and characteristics produced by created and restored wetlands differed from those performed and produced by reference wetlands, except water quality functions, which appeared to be performed in a similar capacity. None of the studies compared the functions provided by compensation wetlands with the functions provided by the wetlands that were lost.

For the most part, reference wetlands provided habitat for a greater diversity or abundance of wildlife than created or restored wetlands. Birds were an exception since half of the studies found no difference between created/restored sites and reference wetlands, particularly for ducks. Created and restored wetlands have different vegetative characteristics and plant communities than reference wetlands. Certain plant communities, such as sedge meadows, may require many years to develop if at all.

The common finding that wetland compensation sites have greater vegetation species richness is probably linked to the broad range of niches created on a new site. A newly created or restored site is a "blank slate" upon which species will be planted, species from the previous habitat on the site will re-emerge, and species adapted to disturbance will colonize. Over time the site will stabilize and mature and only the species adapted to the resulting conditions will remain. However, research on restored, created, or enhanced sites that have stabilized is currently lacking. This indicates that sites are probably not studied for a long enough time, due either to the relatively short regulatory timeframe or the decades or lifetimes necessary to achieve stabilization and maturity.

Researchers observed that created, restored, and enhanced wetlands had less organic matter than reference wetlands. This could be due to the excavation of surface soil layers during project construction. Studies also indicated that organic matter at compensation wetlands did not appear to accumulate over time. Therefore, plant establishment at compensation wetlands could be hindered by low organic content in conjunction with soils that were found to be sandier, more compacted, and lower in nitrogen.

Compensatory mitigation is producing more acreage of open water wetlands than was lost. The ability of compensatory mitigation projects to produce other Cowardin classes varied. Some compensatory mitigation wetlands have produced significantly different HGM classes than were present in the reference wetlands. This has resulted in wetlands that have more inundation for a longer period than reference systems.

Some unique types of wetlands, such as bogs, fens, and mature forested wetlands, may not be reproducible, especially not within current regulatory timeframes. Other wetland types, such as vernal pools, may be reproducible given the right conditions.

The literature provided numerous suggestions on virtually every aspect of the mitigation process. Key suggestions include:

- Improving regulatory guidance on a variety of topics, such as measurable, meaningful, and enforceable performance standards for compensatory mitigation
- Finding better sites that provide increased benefits due to their location within a watershed
- Monitoring compensatory mitigation wetlands more effectively
- Implementing measures to increase regulatory follow-up of compensation projects

The literature suggests that some improvements have been made in compensatory mitigation over the past two decades, particularly in terms of what is required. However, overall success and permit compliance have not noticeably improved. Most studies indicate that created and restored wetlands do not provide the same characteristics or level of functions as reference wetlands (water quality functions may be the exception). Though older created and restored wetlands generally exhibit vegetation characteristics that lead to improved habitat for wildlife, soils and hydroperiods may remain so modified that they will not replicate reference systems in the foreseeable future. Since the effectiveness of compensatory mitigation remains highly variable and somewhat questionable, it is increasingly important to understand the cumulative effects of the continuing loss of wetland acreage and functions. This will be addressed in the next chapter.

Chapter 7
Cumulative Impacts on Wetlands

7.1 Reader's Guide to This Chapter

The synthesis of the scientific literature in Chapters 2, 3, and 4 has clearly established that wetlands do not function in isolation from the landscape that surrounds them. Their ability to provide certain processes and functions is affected by surrounding conditions and land uses within their contributing basins, as well as hydrologic patterns created by changing land uses.

In Chapters 5 and 6, management tools that are applied on a project-by-project basis were discussed relative to the findings of their effectiveness based on the scientific literature. The literature described in previous chapters demonstrates that project-by-project decisions cannot, by their very site-specific nature, adequately address the complexities of wetland systems as they function in a landscape context.

The findings in the scientific literature point toward the need for a broader perspective when considering management options that address, to the extent possible, the myriad of processes that determine whether wetlands remain viable on the landscape. This broader perspective is the subject of this chapter.

7.1.1 Chapter Contents

Major sections of this chapter and the topics they cover include:

Section 7.2, Introduction and Background on Wetland Loss and Cumulative Impacts describes wetland loss in Washington and three studies in the Pacific Northwest that illustrate more recent loss. It describes cumulative impacts and provides an analogy.

Section 7.3, Implications of Current Management Practices for Cumulative Impacts describes how the current wetland management approach of addressing resources parcel by parcel, with variation between jurisdictions, can lead to inadequate protection of wetlands.

Sections 7.4, Cumulative Impacts Are Difficult to Assess discusses some reasons given by researchers for the difficulty in assessing cumulative impacts.

Section 7.5 Assessing and Planning for Cumulative Impacts summarizes the recommendations by researchers for ways to manage aquatic resources on a landscape level rather than case by case. Watershed planning, assessment and selection of restoration sites, floodplain restoration, and other topics are discussed.

Section 7.6, Chapter Summary and Conclusions ties together the major concepts presented in the chapter.

7.1.2 Where to Find Summary Information and Conclusions

There are two brief summaries of the major points resulting from the literature review in a bullet list format. The first covers those sections which precede it, while the second is a summary of the section in which it is located. The reader is encouraged to remember that a review of all sections preceding the summary is necessary for an in-depth understanding of the topic.

For summaries of the information presented in this chapter, see the following sections:

- Section 7.4.1
- Section 7.5.3

In addition, Section 7.6 provides a summary and conclusions about the overarching themes gleaned from the literature and presented in this chapter.

7.1.3 Data Sources

There is adequate literature on cumulative impacts related to broad landscape-scale processes and much of it is recent. Not all of the literature is specific to wetlands. However, the implications of cumulative impacts are not wetland specific. Research has been conducted regarding the consequences of the permitting processes, as well as from the perspective of ecological systems (i.e., habitat loss, fragmentation, metapopulations, and remnant patch dynamics). Much of the literature is focused on urbanization and forest practices, with less information available on agricultural practices as they relate to cumulative effects.

7.2 Introduction and Background on Wetland Loss and Cumulative Impacts

In the 200-year period previous to the late 1980s the state lost an estimated 31% of its 135 million acres (55 million ha) of wetlands (Dahl 1990). In addition, changes have occurred in all types of natural resources, including forests, shrub and steppe communities, and aquatic resources including wetlands. Increasing population and decisions about land uses in the state have resulted in human activities such as diking,

draining, and agricultural practices impacting wetlands, in addition to the direct loss of wetlands (Department of Natural Resources 1998).

No more current data are available for freshwater wetland losses in Washington. However, on the national level, the loss of wetlands has not stopped, according to a report released by the National Research Council (1995). The U.S. Environmental Protection Agency states that although wetland loss rates are slowing, the United States continues to lose approximately 70,000 to 90,000 acres (28,300 to 36,400 ha) of wetlands on non-federal, rural lands each year (U.S. Environmental Protection Agency 2002).

Three studies in the Pacific Northwest illustrate that the loss of wetlands continues in this region:

- Bell (2002) studied sphagnum-dominated peatlands that were originally mapped by Rigg in the early 1950s in King County. Bell found a 69% loss of these wetlands since 1958. Of 26 sites, six remained relatively undisturbed. Eight showed a decline in acreage and quality of plant communities. Five wetlands are now highly disturbed with no sphagnum moss present. The remaining seven wetlands were either drained or filled. Of the 406 acres (162 ha) present in 1958, only 125 acres (50 ha) remain today. The losses were due to agricultural conversion, development, and peat mining.

- A study of recent losses of wetlands within the Willamette Valley, Oregon, found that from 1981/1982 to 1994 there was a loss of approximately 9,500 acres (3,800 ha) of wetlands, representing approximately a 2.1% loss of wetlands within the Willamette Valley study area. They found that 70% of the loss was attributable to agriculture, 6% was associated with the impacts of urbanization, and 24% was attributable to other unidentified causes. (Bernert et al. 1999)

- A study conducted by Holland et al. (1995) in the greater Portland, Oregon, area found that 40% of the wetlands identified in the National Wetland Inventory of 1981/1982 were missing in 1992. They attributed most of the loss to the impacts of urbanization, yet they still attributed 31% of the losses to agricultural conversion. One conclusion of their study was that small, often isolated wetlands were lost due to decisions regarding single-project permits that did not take into account the overall pattern of wetland loss.

Direct loss of wetland acreage and impacts to wetland functions relating to single permits result in cumulative impacts that extend beyond individual wetlands. Johnston (1994) notes that wetlands, even physically isolated ones, are not isolated from other aquatic resources when viewed in a landscape context. Water and many organisms move freely through the landscape, so that impacts to one wetland can affect other wetlands and terrestrial habitats.

Cumulative impacts have been described by Hemond and Benoit (1988) as follows:

> *Wetlands are frequently subject to multiple impacts over time and/or space; the effects of such multiple impacts may be simply additive, or the total effect*

> *may be more severe than the sum of the effects of the individual impacts alone. Cumulative impact as used here refers to multiple impacts whose effects on the wetland cannot be predicted by simply adding the effects of all the individual impacts.*

In addition to the direct filling, draining, and altering of wetlands, less direct effects on wetlands can still result in cumulative impacts. These effects have been discussed in previous chapters and are caused by changes in land use in a wetland and its surrounding area. Wetlands and the functions they provide have been affected in terms of the distance between wetlands, the connectivity of habitat between them, and their location, distribution, and position within watersheds. As described in other chapters of this document, this affects the dispersal of animals and plants between wetlands and the water quality, flood attenuation, and hydrologic processes wetlands can provide.

Cumulative effects of decisions made project-by-project: An analogy

Understanding the implication of managing wetlands and other aquatic resources at a landscape level, rather than a project-by-project level, may seem abstract given the complexity of how ecosystems function in the landscape. The following analogy is offered to provide an alternative description of cumulative effects. Credit for the following analogy was given to Gosselink and Lee by Preston and Bedford (1988):

> *Imagine a Renaissance mosaic of a mother and child, composed of beautiful tiles of various shapes and colors. As it has aged the mosaic has begun to lose tiles. As managers responsible for the mosaic, we have to determine which of the tiles to preserve and reinforce, which to attempt to restore, and which we will allow to be further damaged or even destroyed. Our objective is to attempt to preserve the highest value for the mosaic. Using a tile-by-tile decision methodology (the project-by-project impacts assessment), each tile would be assessed separately and individually for its intrinsic value. Each decision for a tile would not consider the other nearby tiles, nor even how the tiles fit into the whole image. This strategy would very likely not preserve the image of the mother and child. Yet, it is the image that gives the mosaic its inherent value, not the sum of the individual tiles. If one is to preserve the value of the image, then one needs to be able to determine the relative significance of each individual tile relative to each other tile and to the image as a whole.*

The following section describes how the current approach to managing wetlands can lead to cumulative impacts, discusses the need to consider cumulative impacts when making decisions about wetlands, and identifies some reasons given by researchers for the difficulty in assessing cumulative impacts.

7.3 Implications of Current Management Practices for Cumulative Impacts

Some of the causes of cumulative impacts on wetlands stem from how wetlands are regulated in Washington State, and how case-specific decisions are made for each individual project. These findings relative to regulations at the local (city or county) level are discussed further below.

Decisions about wetland protection and management in Washington State occur at the federal, state, and local levels. While federal and state agencies regulate many direct impacts to most wetlands, they do not regulate land uses outside of wetlands and, thus, cannot protect many of the wetland functions. The primary responsibility for protecting wetland functions lies with local governments. In Washington State, local jurisdictions have zoning codes that establish the regulatory framework for wetlands. Thus each city or county has its own independent wetland regulations. In an effort to encourage concurrency between adjacent jurisdictions for wetland planning and implementation, the state legislature passed the Growth Management Act (GMA) in 1992.

The GMA encouraged consistency between adjacent jurisdictions to reduce confusion and conflicts between regulatory standards from one jurisdiction to the next. However, consistency has not always been achieved, and adjacent jurisdictions may have quite different regulatory standards.

On the simplest level, adjacent jurisdictions may have different management objectives for resources and distinctly different criteria and standards. For example, differences in rating systems between two contiguous jurisdictions may result in different ratings for the same large wetland complex that crosses the boundaries between the jurisdictions. Differences in ratings may result in distinctions in protection such as buffer dimensions or replacement ratios. Whether different portions of the same wetland are protected at all could vary from one local jurisdiction to the next.

As noted by Albrecht et al. (1995), delegating authority to counties and municipalities can lead to tremendous variation in land use patterns. The literature is clear that when broad-scale environmental landscape issues, like managing watersheds and ecological processes, require cross-jurisdictional cooperation and collaboration, “Jurisdictional fragmentation impedes wise environmental planning” (Albrecht et al. 1995). Thus, one source of cumulative impacts is the lack of consistent regulations from one jurisdiction to the next for ecological systems that depend upon landscape-scale processes across jurisdictional boundaries.

Although there is currently no substantial literature on the implications of different management strategies between jurisdictions, there is significant literature on the relation between case-by-case decision-making and cumulative impacts. In fact, one reason often cited for the failure of site-specific management to adequately protect aquatic resources is the inability of such an approach to address cumulative impacts (Johnston et al. 1990, U.S. Environmental Protection Agency 1999, Dale et al. 2000).

Research from many sources has clearly identified that piecemeal (case-by-case) implementation of environmental regulations is having a substantial cumulative impact because it fails to identify and account for the landscape-scale processes that create and maintain wetlands (Wissmar and Beschta 1998).

Preston and Bedford (1988) note that making project-by-project decisions fails to evaluate the project and its potential impacts within the spatial and temporal scale within which ecosystems function. They state that although making project-by-project decisions

> *. . . allows evaluation of the local impacts on resources, it does not allow evaluation of impacts of the project on these resources as a whole, of the total impact on these resources from all anthropogenic disturbances, or of secondary impacts resulting from the interaction of impacts from the project with other anthropogenic disturbances. This is true because the spatial and temporal boundaries of the analysis have not fully enclosed spatial and temporal dynamics of the environmental resources of concern and the anthropogenic activities influencing them.*

They recognize that impacts can be generated not only from project-specific actions, but also from actions that occur out of time and outside the vicinity of the activity that may be under scrutiny for a particular project.

Other scientists, such as Everard (1999), are concerned that regulating wetlands and other aquatic resources on a case-by-case basis, without considering watershed or landscape processes, creates the illusion that the resources are being protected by case-by-case management decisions. This has serious ramifications:

1. The public assumes that current land use regulations and management decisions are adequate to protect aquatic systems.

2. There is a resultant public perception that protection of aquatic resources is a continuing financial burden.

If it is assumed by the public, the decision-making bodies (councils, commissions, and/or planning boards), and the implementing staff (planners) that case-by-case permitting is adequate to protect the resources, then there is no incentive to assess or modify the existing policies or regulatory programs. One element that the current scientific research has not adequately addressed is the lack of understanding that the regulatory programs currently in place may not be effective and may be inadvertently deceiving the public regarding protection of complex resources. This is partially due to the difficulty in assessing cumulative effects.

7.4 Cumulative Impacts are Difficult to Assess

In the late 1980s, Bedford and Preston (1988) observed, “The incongruity between the regional scales at which wetland losses are occurring and the project-specific scale at which wetlands are regulated, and also studied, has become obvious.” They pointed out

that research conducted and information generated on a wetland-by-wetland basis does not provide the data necessary to address the complexity of managing ecosystems across a landscape scale. They noted that:

> *…improving the scientific basis for regulation will not come merely from acquiring more information on more variables. It will come from recognizing that a perceptual shift in temporal, spatial, and organizational scale is overdue, the shift in scale will dictate different—not necessarily more—variables to be measured in future wetland research and considered in wetland regulation.*

However, the recommendations from the literature are consistent: Cumulative impacts should be assessed (Johnston 1994, Stein 2001). Abbruzzese et al. (1990) state that once cumulative impact assessment is conducted it "provides a context for wetland permitting." This means that when potential cumulative impacts have been analyzed and identified, then the potential implications of a proposed single action can be assessed in the context of the entire watershed, by assessing historic actions, existing conditions, and watershed configuration and functions.

Assessing cumulative impacts is challenging. The technical issues that complicate an analysis of cumulative impacts include:

- The large spatial and temporal scales involved
- The wide variety of ecological processes and interactions that are present in natural systems
- The lag times that can often separate a land use activity from resulting effects

In addition to the scale of space and time, and the complexity of ecological processes, it is also difficult to differentiate between human-caused and natural variations. It is challenging to discern cause and affect relationships when looking at ecosystems that are subject to both variations in natural processes and associated disturbances, as well as anthropogenic changes. Natural systems are in a constant state of change, responding to landscape variables that influence ecosystem processes. It is difficult in some instances for research to unravel the influence of human-caused disturbances that are embedded within natural variability.

Researchers also recognize that it is difficult to isolate cause-effect relationships in a landscape context. As mentioned previously, Johnston (1994) notes that wetlands, even physically isolated ones, are not isolated from other aquatic resources when viewed in a landscape context. Water and many organisms move freely through the landscape, so that impacts to one wetland can affect other wetlands, terrestrial habitats, and the species that depend upon them. Effects can accumulate in the following ways (Johnston 1994):

- **Time-crowded perturbations**, in which disturbances occur so close together in time that the system cannot recover in the time between.

Example: Reoccurring flooding that drowns vegetation that is not adapted to prolonged inundation.

- **Space-crowded perturbation**, in which disturbances are so closely spaced on the landscape that their effects are not dissipated in the distance between.

  Example: Construction of new highways and high-density commercial zones on both sides of a wetland, resulting in increased noise, lighting, and human presence.

- **Synergisms**, the interaction of disturbances to produce effects that are qualitatively and quantitatively different from the individual disturbances.

  Example: Stormwater discharge into a wetland that causes both a change in the water level fluctuations and an increase in excess nutrients and toxics entering the wetland.

- **Nibbling**, disturbances that produce effects by small incremental changes.

  Example: A 50-foot buffer around a wetland within a residential development in which the homeowners encroach on the buffer over time by expanding their lawns.

- **Indirect effects**, in which disturbances produce effects remote in time or space from the original disturbance.

  Example: Conversion of the upper portion of a watershed into high-density impervious areas, resulting in interception of shallow groundwater recharge, increased surface water volumes, and greater rates of discharge into the natural drainage system.

From Johnston's (1994) findings one can conclude that impacts to wetlands cannot be determined by looking at one point in time or viewing impacts isolated from other actions that have occurred within the landscape historically or may occur in the future.

Inter-jurisdictional decision-making adds another layer of complexity to the issue of cumulative effects. Permit decisions made by one jurisdiction within a watershed can add to the consequences of another jurisdiction's decisions within the same watershed.

The authors cited are in agreement that the consequences of cumulative impacts are easier to assess than the causes. Small incremental changes may not singly be the cause of a major change—it is the synergistic and compounding consequences that may result in the observable cumulative impacts.

Therefore, evaluation of cumulative effects requires a non-traditional way of evaluating resource impacts. As noted by Preston and Bedford (1988):

> *Cumulative effects on freshwater wetland ecosystems are more tangible than any scientific basis for measuring these effects. The notion that individually insignificant actions can produce major change through the accumulation of effects is compelling enough to have influenced federal legislation, initiated court action, and produced international meetings. Yet constraints remain more obvious than any specified approach or method for implementing this idea in natural resource regulation and management.*

"Cumulative effects analysis requires a non-traditional approach to information by recognizing that patterns are more important than details, an interdisciplinary focus is more important than a mono-disciplinary foci, and large areas are more than the sum of their parts and so must be evaluated as a unit" (Reid 1998).

7.4.1 Summary of Key Points

- In Washington State, decisions about wetland protection and management occur at the federal, state, and local levels. Therefore, the same wetland may be subject to a variety of policies and regulatory standards. Permit decisions are most often made on a case-by-case basis. Decisions made on a case-by-case basis by jurisdictions with differing standards can result in cumulative effects and loss of wetland function across the landscape.
- Cumulative effects are difficult to assess because of the large spatial and temporal scales involved, the wide variety of processes and interactions, and the lag times that can often separate a land use activity from resulting effects. Therefore, evaluation of cumulative effects requires a non-traditional way of evaluating resource impacts.
- The causes of cumulative effects are not limited to the policies and regulations of a single agency but can also be caused by the compounding effects of multiple agencies making land use decisions in isolation and without comprehension of the implications of accumulated effects.

7.5 Assessing and Planning for Cumulative Impacts

Despite the difficulties associated with assessing cumulative impacts, several researchers have attempted to create methods for analyzing them on a landscape scale that incorporates historic, present, and proposed actions. In synthesizing the scientific information for assessing cumulative effects, Bedford and Preston (1988) identified that there was sufficient knowledge to prepare quantitative models for hydrology and water quality. Brinson (1988) called for assessing the location and condition of wetlands within a watershed or landscape as one means to ensure that long-term cumulative effects on water quality are addressed. Brinson notes that under normal conditions, wetlands tend to function as sinks for nutrients, metals, and organic litter, with modest accretion rates

over time. However, when the wetland hydroperiod is changed, wetlands can become significant exporters of accumulated nutrients, toxics, or sediments through subsidence and export, a decrease in performance of water quality functions due to structural change in the wetland, thereby having potential adverse effects on aquatic systems downstream. This information can be integrated into a landscape-scale management or prioritization plan.

Stakhiv (1988) states that cumulative impact analysis cannot be limited to the effects on ecological processes. The author provides model formulas for quantifying the consequences of project-by-project decisions by incorporating the human social and economic consequences as well. He cautions that analysis of cumulative impacts needs this broader perspective to create and maintain validity in the regulatory context.

All of the preceding authors present strong science-based rationales for analyzing existing conditions at a watershed scale or a landscape scale that may be smaller than an entire basin. However, most of these authors do not address the difficulties of conducting landscape-scale assessments, or the inherent political challenges of conducting such assessments across jurisdictional boundaries.

7.5.1 Planning Approaches Using a Landscape Perspective

Advanced planning is required to minimize cumulative effects and restore systems that have already been impacted. This section provides examples of approaches and planning efforts that incorporate a landscape approach, whereas the next section presents examples of restoration efforts. Much of the available literature on watershed or landscape planning is in the context of streams and rivers rather than wetlands, with the exception of the West Eugene Plan. However, the findings and conclusions of that literature were determined to be relevant to wetland planning.

Stein (2001) presents an approach to planning based on placing aquatic systems in the context of the greater landscape, including the floodplain, riparian corridors, and connectivity of processes. Stein's recommendations include identifying areas that provide functions at a higher level early, in order to preserve them at the "cost" of allowing the loss of areas with lower levels of function. It is not possible to attempt such a trade-off in a situation where projects are analyzed on a case-by-case basis and no landscape-level analysis has been conducted.

It may be hypothesized, however, that this approach may result in the loss of wetlands whose restoration might be important for improving processes and functions at the landscape level. Some degraded wetlands that are currently not performing functions well may be in a critical position in the landscape and are important locations for restoring process throughout a watershed. For example, there are currently many degraded wetlands behind dikes at the mouth of the Snohomish River. These wetlands, however, can easily be restored to provide significant habitat for salmonids (City of Everett 1997). If these wetlands are lost, the potential for restoring salmonid habitat in the watershed is reduced.

Stein also calls for understanding past decisions that have been made and the consequences of those decisions relative to the ecological processes that continue within the landscape scale. That is only possible with a landscape-scale analysis prior to making decisions regarding single projects.

Stein recommends:

- Anticipating project-specific impacts in the context of past actions and the landscape-level processes in the project area
- Maintaining floodplain integrity by considering the impacts to the entire riparian zone during the planning process and permit review
- Using wider, more natural riparian corridors to stabilize banks, minimize erosion, and reduce flow velocities
- Creating permit and mitigation requirements to reflect the magnitude and permanence of the impact
- Preserving portions of the landscape that have higher function or are more valued as compensation for loss of lower functioning portions

Another approach recommended by Schueler and Holland (2000) involves applying zoning at a watershed scale. They propose a classification scheme for streams in urban and urbanizing areas with specific resource objectives for each stream and subwatershed. "Specific policies and practices [that address elements such as] impervious cover limits, stormwater practices, and buffers are then instituted to meet the stream resource objective, and these practices directly applied to future development projects."

Schueler and Holland conclude that the process of planning at the landscape scale "forces local governments to make hard choices about which streams will be fully protected and which will become at least partially degraded. Some environmentalist and regulators will be justifiably concerned about the streams whose quality is explicitly sacrificed under this scheme." They conclude that when a jurisdiction looks at the entire aquatic system within a watershed, rather than the permit-by-permit approach, the jurisdiction can generate more accurate information and make more informed choices about how the entire aquatic system works, recognizing the consequences of difficult choices up front.

They acknowledge that this will result in further impacting some lower priority resources while allowing land use changes to occur in a more informed manner. Their suggested approach, as with that of Stein (2001), is based on the premise of identifying and assessing an entire watershed and/or landscape in order to understand the processes that are linked, and prioritize choices about resource protection. This approach also implies that decision-makers and the public are more thoroughly informed as to the consequences of the choices that are made.

The West Eugene Wetlands Special Areas Study or "West Eugene Plan," prepared for West Eugene, Oregon, is a well known and documented, multi-objective, watershed-based planning effort. Gordon (1995) describes the plan as being implemented at the

local level to address issues of wetland and wildlife habitat protection, flood control, stormwater quality treatment, recreation and education, as well as research. He notes that, "Landscape approaches allow systems to be viewed from a natural and human perspective. A comprehensive approach allows environmental and urban development considerations to be viewed together."

The West Eugene Plan assessed all the resources within a planning area covering 16 square miles where there is a large concentration of wetlands. It considered multiple variables including economic considerations, infrastructure, and restoration potential. Through an intense collaborative process the plan has been adopted and implemented by the city. It provides a model for how prioritized, watershed-based planning can be adopted and implemented in an urban watershed.

Weber and Wolf (2000) describe a program in the State of Maryland where the state and local agencies are working together on a landscape scale to create an interconnected corridor network of prioritized critical habitats and linkages based on ecological parameters and threat parameters. This statewide process, the Green Infrastructure Assessment (GIA), is based on identifying hubs and corridors using a geographic information system (GIS). It identifies hubs on two tiers: statewide significant hubs must be 2,000 acres (809 ha), and hubs of local concern are 500 to 2,000 acres (202 to 809 ha). Corridors link the statewide or local hubs and include "nodes," which can be patches of interior forest, wetlands, habitat areas for identified sensitive species, or other protected areas located along the corridors. Although this particular program is centered on "green infrastructure" (not just aquatic resources), their findings are clearly germane to any landscape-scale management or protection program.

Their approach was based on the following:

- The role of a given place as part of a larger interconnected ecological system
- The value of integrating the interests of multiple resources into a single framework
- The importance of considering the integrity of natural resources/ecosystems in the context of existing and potential human impacts to the landscape
- The importance of regional (i.e., inter-jurisdictional) coordination of local planning
- The need for a regional element to a strategy for the conservation of biodiversity

This approach recognizes the need to incorporate coordination with regional and local planning efforts along with existing and potential human impacts. It is a pragmatic approach that incorporates input from all levels, recognizes that humans are a part of the problem and the solution, and realizes that the solution must reflect regional and local planning and implementation frameworks.

7.5.2 Landscape-Scale Planning and Restoration

Planning for restoration within a watershed also requires an analysis of cumulative effects and landscape-scale processes. The literature contains consistent recommendations for incorporating cumulative impact analysis into the process of undertaking watershed restoration actions.

In writing about restoration of riparian ecosystems, Wissmar and Beschta (1998) make it clear that the landscape context, including all the historic, ongoing, and future influences on an ecosystem, must be considered when attempting restoration actions. This same broad perspective is required in the beginning of the planning process. If the goal is to maintain sustainable ecosystems, then one has to identify and understand how they function within the landscape and what elements must be accounted for to ensure long-term viability.

Bedford (1999) argues that one should conduct a cumulative impact analysis for the region when considering wetland restoration activities in a watershed. The analysis should not be focused on the amount of historic wetland loss but rather on the concept of "templates for wetland development" that control the "hydrologic variables and hydrologically influenced chemical variables" that form and maintain wetlands. This is necessary in order to make informed decisions about the suitability of specific restoration actions in particular landscape positions.

Bedford goes on to state:

> *. . . wetland restoration has the potential to increase wetland diversity as well as wetland area but only if those planning restorations do so within the context of cumulative effects analysis that considers landscape patterns of loss and degradation and landscape controls of wetland development. A cumulative effects analysis draws attention to wetlands as landscape elements.*

Smith and Jones (1997) describe the need to take a landscape approach to restoration and protection of riparian communities with an understanding of the relationships between geomorphology, hydrology, and vegetation. Their paper outlines a two-stage approach to prioritizing restoration at two different scales, the reach scale and the site scale. This approach is again based on the concept that landscape processes beyond the site scale are vital to design and decision-making.

The Drayton Harbor, Washington, sub-basin analysis (Stanley et al. 2003) is an example of a landscape approach that identifies restorable wetland sites based on recovery of watershed processes. This analysis targeted wetlands for restoration based on their potential for restoring functions, structure, and processes that address environmental problems at the landscape scale. Using this approach, deficient functions within a sub-watershed can be addressed such as water quality improvement, flood attenuation, habitat losses, etc. This method is a flexible tool for tailoring wetland restoration choices to the most desirable location for achieving the greatest benefits to functions.

Almendinger (1999) also discusses a method of prioritizing wetland restoration to achieve water quality improvement in agricultural basins, with an example in the Minnesota River Basin. The author categorizes three factors that affect water quality improvement in wetlands: loading factors, path factors, and process factors. These factors affect the amount, timing, and chemical alteration of suspended sediments and nutrients.

Three key questions should be asked about site selection factors that will affect the wetland's ability to improve water quality:

- Is the site in an area with known water quality problems? (problem effectiveness)
- Is the site likely to improve water quality more or less than other sites? (function effectiveness)
- Does the site fit within an overall research plan to gain information on how wetlands improve water quality? (information effectiveness)

In addition to water quality benefits, Almendinger (1999) discusses ways to integrate other benefits of restoration, including wildlife habitat and flood attenuation. The author concludes:

> *Re-conversion of productive agricultural land to wetland will be expensive, but should be considered in light of the original landscape, and the cost of doing nothing to change the present landscape. Pimentel et al. (1995) argue that the cost due to soil erosion under current practices is about five times the cost needed to implement soil conserving practices such as upland BMP's and wetland restoration. In the long run, wetland restoration may be far less expensive than doing nothing.*

This guidance is applicable to restoration efforts at the landscape scale across the country.

A few authors focused on planning for restoration of floodplains, riparian areas, and associated aquatic habitats using a landscape approach. Most large river floodplains in the Pacific Northwest once supported extensive wetland systems. Years of agricultural conversion by diking, dredging, port development, and urbanization have resulted in significant loss of wetlands in the lower floodplains, 90 to 99% in most systems (Bortelson et al. 1980). Thus, the restoration of floodplain and riparian wetlands offers an opportunity to address these losses as well as other issues.

Coulton et al. (1996) studied the Willamette River, Oregon, to assist the U.S. Army Corps of Engineers in modifying current practices toward an integrated long-range plan for restoring the flood capacity of the watershed, its aquatic and riparian habitat, water quality, and threatened and endangered species. The report identified several indirect benefits from restoring former wetlands in the floodplain:

- Increasing seasonal water availability
- Reducing downstream sediment

- Improving water quality

The report recommends using a hydrology database and a geomorphology model for the river as part of the method to prioritize and select sites for restoration. A synthesis of the literature on the Willamette River specifically, and watershed and stream restoration generally (particularly in urban areas), is available from Portland Metro (2002).

Goodwin et al. (1997) found over 400 papers published between 1970 and 1995 on the subject of riparian restoration. They recommend looking at processes within the watershed and responding to overarching problems—not local symptoms—when approaching riparian restoration. The principles presented in their paper are equally applicable to restoration of riparian wetlands. They suggest asking the following regarding the causes of degradation:

- Is the disturbance local or does it originate outside the degraded system?
- Is the disturbance ongoing or can it be eliminated?
- Will the system recover on its own if the disturbance is removed?

Hawkins et al. (1997) found that planning restoration and selecting sites in riparian areas (for compensatory mitigation or on a voluntary basis) must include an evaluation of the vulnerability of the sites to disturbances caused in other locations in the landscape. They argue that it is not enough to select sites that have the highest potential for supporting wetland functions: It is necessary to couple the selection criteria with an evaluation of the vulnerability of different sites to natural and anthropogenic disturbances. They found that flooding damage to riparian vegetation was related to the amount of riparian vegetation present in the floodplain and the amount of urban surface in the upstream watershed. They conclude:

> *. . . sites near existing large areas of intact riparian vegetation and away from urban development will have the highest potential for successful long-term restoration . . . The key to selecting sites with the greatest overall potential for successful restoration is to properly weigh different site selection criteria in context to overall restoration objectives (e.g., what riparian values and functions are most desirable).*

The need to incorporate the degree of disturbance in the watershed is equally relevant to the Pacific Northwest and to compensatory mitigation. A case in point is a wetland that was created to compensate for the impacts of the Nisqually Fish Hatchery on Clear Creek. Soon after the riverine site met its regulatory performance standards, it was damaged during a flood event. The 1996 flood deposited up to 5 feet (1.5 m) of alluvial material in the outflow channel constructed for the created wetland. The material buried vegetation and separated the wetland from the river, converting it to a depressional wetland (Wiltermood, personal communication). Caution is recommended when conducting wetland projects in highly dynamic landscape settings.

7.5.3 Summary of Key Points

- Project-by-project permits do not address the complexities of wetland and ecosystem processes that the scientific literature has clearly identified. Such case-by-case decision-making results in cumulative impacts and jeopardizes the long-term sustainability of aquatic resources.

- Researchers have suggested assessment methods and planning approaches for use in addressing cumulative impacts to aquatic resources. Elements include considering an entire watershed, recognizing past land use patterns and thereby anticipating project-specific actions, preserving those portions of the landscape with higher or more valuable functions as compensation for loss of lower functioning portions, and prioritizing the resources or goals and objectives as a basis for the management plan. Some authors have included social and economic considerations within the formulation of a watershed or landscape plan.

- The literature recommends a watershed approach for the management and restoration of aquatic communities. Researchers recognize the need for an analysis of cumulative effects (the historic, ongoing, and future impacts on an ecosystem) and landscape-scale process (including geomorphology, hydrology, vegetation, etc). The most desirable locations for restoration can then be chosen in order to achieve the greatest benefits.

- It is not enough to select restoration sites that have the highest potential for supporting wetland functions. The selection criteria need to be coupled with an evaluation of the vulnerability of different sites to both natural and human disturbances within a landscape context. This is the only way to ensure long-term success of the restoration.

- Several landscape-scale planning efforts have been implemented in the Pacific Northwest region. These provide local models for understanding approaches, consequences, and strategies for implementation.

7.6 Chapter Summary and Conclusions

The majority of wetland management decisions in Washington State are based on case-by-case actions related to specific projects, without any opportunity to consider landscape-scale processes or consequences. This pattern is a result of the current structure of programs at local, state, and federal regulatory agencies. The results of the research on case-by-case permitting processes are clear: There are consistent wetland losses regionally and statewide.

In spite of regulatory programs at federal, state, and local levels, wetland impacts continue to occur. These impacts are often the result of cumulative and synergistic impacts across the landscape.

The benefit and anticipated consequences of managing natural resources within a landscape context are well documented. In the case of wetland resources, a landscape-scale management program has not been implemented in Washington State. However, examples from Oregon and elsewhere in the nation provide approaches that may help address cumulative impacts in Washington. The literature reviewed for this synthesis did not focus on the reasons for the lack of landscape-scale strategies, but some impediments can be assumed:

- Costs of analysis, inventories, assessments, and rankings
- Costs of implementing a landscape-scale program relative to existing project-driven programs that are often funded by applicant fees
- Inconsistent mandates driving the agendas and priorities of regulatory agencies
- Lack of examples of successful tools for interagency collaboration and implementation
- Lack of awareness and understanding of the ecological consequences of existing regulatory programs by the public and the staff of implementing agencies
- Lack of support for local jurisdictions to tackle the process of identifying and prioritizing aquatic resources for long-term protection and/or potential alteration

Volume 2 of this two-volume document will build on the foundation of scientific information in this synthesis to construct options and recommendations for the protection and management of wetlands and their functions at the site and landscape scales. The production of Volume 2 will begin during the review period for Volume 1.

References

The following list of references includes those literature sources obtained, reviewed, and synthesized by the Core Team that prepared this report. The list also includes references cited in those portions of *Indicators for Monitoring Biological Integrity of Inland, Freshwater Wetlands* (Adamus et al. 2001) that were adapted for use in this synthesis. The references from the adapted portions of the Adamus et al. (2001) document have not all been obtained and reviewed by the Core Team, but they are included here for the convenience of the reader who wishes to consult the original literature for further information.

In reviewing the text, we have identified approximately 9 citations for which we don't have all reference information. These citations will be provided in the final draft.

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Glossary

Adventitious roots. Additional roots that develop in some plants, such as willows and alders, as an adaptation to saturated or flooded conditions.

Bog. A unique type of wetland dominated by mosses that form organic peat. Bogs form in areas where the climate allows the accumulation of peat to exceed its decomposition. Bog hydrology is dominated by precipitation rather than surface inflow. The plant community is specialized to survive in the nutrient-poor and highly acidic conditions typical of bog systems.

Canopy cover. The degree to which the foliage of the highest vegetation layer in a plant community blocks sunlight or obscures the sky.

Capillary action. The movement of water through tiny spaces as a result of cohesion (water molecules sticking to other water molecules) and adhesion (water molecules sticking to molecules of other substances). Capillary action can result in the soil being saturated for several inches above the water table in some soil types, as water molecules move upward through tiny pores in the soil.

Capillary fringe. The area immediately above the water table in which water is drawn upward through the soil by capillary action.

Conductivity. A measure of the amount of dissolved constituents (ions) in water, based on the water's ability to conduct electricity. See *specific conductance*.

Contributing basin. An area from which surface water drains to a particular wetland.

Corixids. A group of aquatic insects commonly called "water boatmen."

Cowardin classification. The first commonly used classification system for wetlands developed in 1979 by the U.S. Fish and Wildlife Service. The Cowardin system classifies wetlands based on water flow, substrate types, vegetation types, and dominant plant species.

Cryptogamic crust. A thin crust composed of mosses, lichens, algae, and bacteria that forms in areas of bare ground between shrubs, grasses, and other plants. Cryptogamic crust is found in undisturbed arid and semi-arid lands such as the shrub-steppe ecosystem of the Columbia Basin.

Decomposer. An organism that breaks down organic material into smaller particles. The action of decomposers allows the chemical constituents of organic matter to become available for uptake by plants, thus "recycling" this material back into the food chain.

Dioxin. A group of several hundred chemical compounds that share certain chemical structures and biological characteristics. They include the chlorinated dibenzo-*p*-dioxins (CDDs), chlorinated dibenzofurans (CDFs), and some polychlorinated

biphenyls (PCBs). The term dioxin is also used to refer to a well studied and toxic dioxin, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD).

Dytiscids. Predaceous diving beetles.

Ecoregion. Geographic regions where climatic conditions are similar and the ecosystems (including wetlands) are relatively homogeneous.

Ecotone. An area that is transitional between two different types of ecosystems and has some of the features of both. Wetlands are often characterized as being ecotones between aquatic and terrestrial ecosystems.

Edge. The boundary where habitats meet or where successional stages of plant communities come together; for example, between an open wet meadow and a mature forest.

Emergence trap. A device placed over the water or sediment in a wetland to capture flying aquatic insects as they emerge from their non-flying larval state into their winged adult form.

Eutrophication. Enrichment of a water body by nutrients, often as a result of human activities.

Evapotranspiration. The combination of water that is evaporated from the surface and that is transpired from the leaves of plants as part of their metabolic process.

Fen. A type of wetland similar to a bog, containing accumulated peat. Fens support marsh-like vegetation including sedges and wildflowers. Fens differ from bogs in their plant communities, hydrology, and water chemistry. They are fed by groundwater and are not as acidic as bogs.

Forb. Any herbaceous plant that is not a grass or sedge.

Functional feeding group. A group of animals (aquatic insects, birds, etc) that feed in a similar way. For example, insects that scrape algae from rocks in a stream are called scrapers; those that shred leaf material are caked shredders; and those that filter small particles from the water column are filter feeders.

Furans. A chemical substance resulting from the manufacture of organic compounds such as nylon.

Geomorphology. The geologic composition and structure of a landscape—its topography, landforms, soils, and geology.

Hemipterans. A group of insects with straw-like, sucking mouth parts.

Herbaceous stratum. A layer of non-woody vegetation, usually less than 6 feet (2 m) tall.

Hertz (Hz). A unit of frequency equal to one cycle per second.

Hortonian flow. A phenomenon that results when the intensity of rainfall exceeds the capacity of the soil to absorb it, and water runs across the surface.

Humic. Of or pertaining to humus, which consists of partially or wholly decayed plant matter.

Hydrogeomorphic (HGM) classification. A system used to classify wetlands based on the position of the wetland in the landscape (geomorphic setting), the water source for the wetland, and the flow and fluctuation of the water once in the wetland.

Hydroperiod. The pattern of water level fluctuations in a wetland. Includes the depth, frequency, duration, and timing of inundation or flooding. Patterns can be daily, monthly, seasonal, annual or longer term.

Interior species. Animal species that require the conditions found on the interior of a habitat type and which are subject to disturbance in areas toward the edges of that habitat. For example, forest interior birds find optimum conditions within the center of a forested area where they are not subject to domestic pets, noise, severe weather, or other disturbances that penetrate the outer forest edge.

Jurisdictional wetland. A wetland that is regulated by the provisions of the law under the jurisdiction of one or more federal, state, or local agencies. Not all areas of the landscape that have the biological characteristics of wetlands are regulated or jurisdictional wetlands.

Lacustrine. Pertaining to lakes or lake shores.

Large woody debris (LWD). Large pieces of downed wood such as logs, rootwads, and limbs that are in or near a body of water. LWD provides habitat structure for fish and other aquatic organisms.

Lentic. Having slow moving or still water, such as a pond or lake.

Lotic. Having running water, such as a river or stream.

Macrophytes. Plants that can be seen with the unaided eye. This includes all vascular plant species, mosses, and large algae.

Metapopulation. A group of local populations between which individuals can migrate.

Microbe. A microscopic organism, such as a bacterium.

Microhm. A unit of measure describing the resistance of a substance to electrical current.

MilliSiemens. A unit of measure for conductivity. See *specific conductivity*.

Mycorrhizae. The symbiotic association of a fungus with the roots of a plant.

Niche. The area within a habitat occupied by an organism; the set of functional relationships of an organism or population to the environment it occupies.

Oligotrophic. Water that is relatively low in nutrients.

PCBs. Polychlorinated biphenyls, a type of toxic chemical compound once widely used in electrical equipment. See *dioxin*.

Periphyton. Plants growing on the bottom substrate of a water body.

Phreatic zone. The area above the groundwater table.

Recruitment (of woody debris). The movement of large and small wood from surrounding areas into an aquatic system over time through the actions of wind, water, or other means. The potential for recruitment of woody debris influences the long-term habitat structure within an aquatic system.

Redox potential. Reduction-oxidation potential, or a measure of the potential movement of electrons in a system. Reduction refers to the chemical process whereby molecules of a substance gain an electron. Oxidation refers to loss of electrons. Measuring the redox potential of a wetland soil provides information about the types of chemical reactions that are occurring in the soil, and thus whether the soil is more aerobic (contains oxygen) or anaerobic (lacks oxygen).

Richness. The number of different species of organisms present in a community.

Riparian. The strip of land adjacent to a body of water that is transitional between the aquatic system and the upland. Some riparian areas contain wetlands.

Rotifers. Minute organisms that live in fresh and salt water. A crown of hair-like structures (cilia) propels them through the water.

Roughness. The amount of friction or resistance a surface provides against water flow. For example, an area containing shrubs and downed branches has greater roughness than a mowed lawn.

Specific conductance. A measure of electrical conductivity standardized to 25°C. Use of specific conductance accounts for the fact that the conductivity of water changes as its temperature changes. It is measured in units of milliSiemens per centimeter.

Stratigraphy. The layers (strata) in a geologic formation.

Subbasin. A smaller drainage basin that is part of a larger drainage basin or watershed. For example, the watershed of a large river may be composed of several subbasins, one for each of the river's tributaries.

Temporal loss of function. The concept that there is a time lag between the loss of existing wetland functions through human or natural disturbance and the reestablishment of functions over time.

Trophic level. A concept used to describe feeding levels in a foodweb. Plants fill the first trophic level by utilizing sunlight to create carbohydrates and other compounds. Plants are consumed by plant-eating animals (herbivores) in the

second trophic level, which in turn become food for predators in the next trophic level, and so on.

Watershed. A geographic area of land bounded by topographic high points in which water drains to a common destination.

Appendix A

Team Guiding Production of Volume 1

An interagency team (the Core Team) guided all aspects of and participated in the search and reading of the scientific literature, wrote the synthesis, and produced Volume 1. The team consisted of staff from the Washington State Department of Ecology, the Washington State Department of Fish and Wildlife, the U.S. Environmental Protection Agency and Sheldon and Associates, the consulting firm hired to assist with production. Additional Ecology staff served as authors (see the list of authors on the title page of this document). The editor was included on the Core Team in the later stages of production.

The Core Team included the following individuals:

Ralph Rodgers	U.S. Environmental Protection Agency
Katherine March	Washington State Department of Fish and Wildlife
Andy McMillan	Washington State Department of Ecology
Tom Hruby	Washington State Department of Ecology
Erik Stockdale	Washington State Department of Ecology
Teri Granger	Washington State Department of Ecology (coordinator)
Dyanne Sheldon	Sheldon and Associates
Kim Harper	Sheldon and Associates
Sara Noland	2N Publications (editor)

Appendix B

Characteristics of a Valid Scientific Process

The characteristics of a valid scientific process in the context of "best available science" are defined below, as quoted directly from WAC 365-195-900:

1. ***Peer review.*** *The information has been critically reviewed by other persons who are qualified scientific experts in that scientific discipline. The criticism of the peer reviewers has been addressed by the proponents of the information. Publication in a refereed scientific journal usually indicates that the information has been appropriately peer-reviewed.*

2. ***Methods.*** *The methods that were used to obtain the information are clearly stated and able to be replicated. The methods are standardized in the pertinent scientific discipline or, if not, the methods have been appropriately peer-reviewed to assure their reliability and validity.*

3. ***Logical conclusions and reasonable inferences.*** *The conclusions presented are based on reasonable assumptions supported by other studies and consistent with the general theory underlying the assumptions. The conclusions are logically and reasonably derived from the assumptions and supported by the data presented. Any gaps in information and inconsistencies with other pertinent scientific information are adequately explained.*

4. ***Quantitative analysis.*** *The data have been analyzed using appropriate statistical or quantitative methods.*

5. ***Context.*** *The information is placed in proper context. The assumptions, analytical techniques, data, and conclusions are appropriately framed with respect to the prevailing body of pertinent scientific knowledge.*

6. ***References.*** *The assumptions, analytical techniques, and conclusions are well referenced with citations to relevant, credible literature and other pertinent existing information.*

Information derived from one of these sources can be considered scientific information if it possesses the required characteristics shown in Table B-1.

Table B-1. Source types and characteristics of scientific information.

Sources of Scientific Information	Characteristics					
	Peer Review	Methods	Logical Conclusions & Reasonable Inferences	Quantitative Analysis	Context	References
A. Research. Research data collected and analyzed as part of a controlled experiment (or other appropriate method) to test a specific hypothesis.	X	X	X	X	X	X
B. Monitoring. Monitoring data collected periodically over time to determine a resource trend or evaluate a management program.	NA	X	X	Y	X	X
C. Inventory. Inventory data collected from an entire population or population segment (e.g., individuals in a plant or animal species) or an entire ecosystem or ecosystem segment (e.g., the species in a particular wetland).	NA	X	X	Y	X	X
D. Survey. Survey data collected from a statistical sample from a population or ecosystem.	NA	X	X	Y	X	X
E. Modeling. Mathematical or symbolic simulation or representation of a natural system. Models generally are used to understand and explain occurrences that cannot be directly observed.	X	X	X	X	X	X
F. Assessment. Inspection and evaluation of site-specific information by a qualified scientific expert. An assessment may or may not involve collection of new data.	NA	X	X	NA	X	X
G. Synthesis. A comprehensive review and explanation of pertinent literature and other relevant existing knowledge by a qualified scientific expert.	X	X	X	NA	X	X
H. Expert Opinion. Statement of a qualified scientific expert based on his or her best professional judgment and experience in the pertinent scientific discipline. The opinion may or may not be based on site-specific information.	NA	NA	X	NA	X	X

X = Characteristic must be present for information derived to be considered scientifically valid and reliable.
Y = Presence of characteristic strengthens scientific validity and reliability of information derived, but is not essential to ensure scientific validity and reliability.
NA = The characteristic does not apply to the source type. For example, monitoring data are not typically peer reviewed.

Appendix C

Methods Used for Searching and Reviewing the Literature

2.1 Searching the Literature

To begin the literature review for Volume 1, personal bibliographies were solicited from a small number of professionals known to have extensive libraries on wetlands in the Pacific Northwest. Other existing published reference lists were reviewed for relevant documents. In addition to the specified reference lists, computer searches were conducted of publicly available databases using a variety of keywords. Table C-1 lists the sources of reference lists and the names of the databases searched, as well as the approximate numbers of documents contained in each source.

Table C-2 lists the keywords that were used in the searches of computer databases. This list was developed by the Core Team and expanded based on comments from focus groups (see Chapter 1 for information on focus groups). The searches were done combining the word "wetland" plus one of the keywords. The words in the last column were used to exclude wetland types not covered by this report. Specific wetland types not found in Washington and known to be very dissimilar from Washington wetlands were also excluded, as were estuarine and marine wetlands.

Table C-1. Summary of reference lists and database searches for Volume 1.

List Source	Approx. No. Documents	Notes
Personal Bibliographies		
Dr. Paul Adamus, EPA	1,600	Broad range of documents
Dr. Tom Hruby, WA Ecology	600	Broad range of documents, many focus on wetland functions
Mary Kentula, EPA	170	Focus on wetland mitigation, management, policy effectiveness
Dr. Klaus Richter, King County	3,500	Focus on amphibians w/Pac. NW emphasis
Published Reference Lists		
Management recommendations for WA priority habitats: freshwater wetlands and fresh deepwater (Morgan 1998)	640	Focus on wildlife and aquatic habitats
Management recommendations for WA priority habitats: riparian (Knutson and Naef 1997)	550	Focus on riparian habitats, not necessarily wetlands
Managing for enhancement of riparian and wetland areas of the Western U.S.: an annotated bibliography (Koehler and Thomas 2000)	1,900	Broad application to western U.S.; many documents not relevant to Pac. NW
Classification and management of aquatic, riparian and wetland sites on the national forests of Eastern Washington (Kovalchik 2001)	400	Focus on eastside and forested areas
Effects of urbanization on pond-breeding amphibians: an annotated literature review (Ostergaard 2000)	100	Focus on amphibians and urban effects
Database Searches		
Keyword searches of various databases	9,800	Databases searched included Ovid, ProQuest, Biosis, Dissertation Abstracts, Agricola, Current Contents, Biological Abstracts
Total	**~17,860**	**Total includes an unknown number of duplicates among the various sources**

Table C-2. Keywords used in searching computer databases of literature.

Base Word	Keywords		Exclusions
Wetland	Aesthetics Agriculture Alkali Alluvial Amphibians Aquifer Recharge Arid Land Artesian Birds Bog Buffers Compensation Conservation Cumulative Impacts Development Disturbed Dynamic Economics Enhancement Erosion Farmed Fen Fish Floodplain Fluvial Functions Geology Geomorphology Grazing Groundwater Habitat Hydraulic Hydric Hydrology Hyporheic Industrial Inventory Invertebrates Irrigation Isolated	Land Use Landscape Maintenance Mammals Mapping Mining Mitigation Mollusks Monitoring Nutrients Perched Policy Public Access Recreation Regulation Reptiles Residential Restoration River Rural Seasonal Septic Slope Soils Spatial Stewardship Stormwater Transportation Corridors Urban Utility Corridors Values Variation Vegetation Types Vernal Pools (not Calif.) Water Quality Water Regime Wells Wildlife	Bottomland Hardwood California Vernal Pools Estuarine Intertidal Lacustrine Marine Mississippi Floodplain Mudflats Salt Marsh Saltwater

2.2 Reviewing, Sorting, and Prioritizing the Reference Lists

Lists resulting from the searches of the computer databases were compiled into a ProCite® database for the project. Documents from other sources found later in the project have been or will be added to the database at a later date.

All reference lists were reviewed by one or more of the Core Team members. From these lists, the Core Team selected those documents that were believed to be relevant to the project, based solely on the title of the article and its date. Those marked documents were then prioritized using a two-tiered system in which those considered most critical to the project were designated as those to be obtained first. Eventually, attempts were made to obtain all the documents on the lists that were believed to be relevant based on their titles. In addition, references were found while individual authors searched for subjects for which information was lacking. These references are provided in the list of references cited in the report but not all been entered into the database at this time.

For the most part, available documents from the past 10 years were used as the primary sources for this report. It was assumed that this most recent literature would incorporate relevant science from the preceding years. Older documents were used in instances where they had not been superseded by more recent studies.

Most of the documents used as sources for Volume 1 meet the criteria for BAS in WAC 365-195-900. The vast majority of the sources were peer reviewed. Conference proceedings and other "gray" literature were occasionally used and in some cases had not been peer reviewed. Peer reviewers are asked to judge the reliability of the sources used, including any gray literature.

In some cases we have cited unpublished data collected by Ecology staff during the calibration of the Washington State Wetland Function Assessment Methods and the State Wetland Rating System. These data have not been published in scientific journals. However, the assessment methods and the wetland rating system which the data support have been peer reviewed.

2.3 Obtaining and Archiving the Documents

Of the more than 17,000 documents on all lists used, copies of over 1,400 documents were obtained after review of the titles and dates, as prioritized using the screening process described above. Paper copies of most of the articles reviewed for this project will be held in an archive at the Washington State Department of Ecology when the project is completed. The archive will be accessible to the public by appointment. A number of theses, dissertations, and books are not included in the archive due to copyright laws and the limited options for purchasing such documents. In these cases, borrowed copies were used and returned, with only the title pages and tables of contents copied for the archive.

2.4 Reading Documents and Writing the Report

References were skimmed and those dealing with Washington or the Pacific Northwest and with practical application to the management and protection of wetlands were prioritized for reading. Searches of the database or the original articles were used by each author to write their portions of the draft document.

Preliminary drafts of Volume 1 were reviewed by the Core Team and selected agency staff, and revisions were made. The revised draft is now being circulated to a group of peer experts as well as anyone who wished to review it. The comments will be compiled and reviewed by the Core Team and further revisions will be made. Volume 1 will be finalized after a draft of Volume 2, providing options and recommendations for managing and protecting freshwater wetlands, is prepared for public review.

Appendix D

Reviewers of Volume 1

This draft of Volume 1 is currently out for review. The final document will contain a list, in this appendix, of all those who provided written comments on the draft.

Appendix E
Methods for Organizing and Grouping Wetland Information

The following information is adapted from Hruby (1999).

Many groups including federal and state agencies have been developing techniques for analyzing wetland functions ever since wetlands were first subject to regulation in the 1970s. The motivation for developing such methods has primarily been the need to predict the effects of alterations to wetlands and set appropriate requirements for compensatory mitigation.

Methods for organizing knowledge about wetlands have been called classifications, categorizations, characterizations, ratings, assessments, and evaluations. These groupings are meant to indicate the type of information a method provides. Unfortunately, the scientific community has been sloppy in the use of these terms to the extent of misnaming many of the analytical tools developed. Users of methods developed for analyzing wetlands should be aware of some of these problems with definitions. Standard definitions for analytical methods based on Webster's Seventh New Collegiate Dictionary (1963) are described below.

Classification/categorization—a systematic grouping into categories according to established criteria or shared characteristics. The two most common wetland classifications are those of Cowardin et al. (1979), which is based on shared characteristics of vegetation and water regime, and the hydrogeomorphic classification (Brinson 1993), which is based on shared characteristics of geomorphic setting and water regime. The criteria used for grouping are generally not linked to specific functions, and thus classifications are not true methods for assessing functions. They can, however, provide a basis on which to develop assessment methods (Brinson 1995).

Characterization—a grouping by a distinguishing trait, quality, or property. For example, the Oregon method (Roth et al. 1993) characterizes wetlands by the properties of "provides" a specific function; "has the potential to provide" a function; or "does not provide" a function. These are three distinct attributes that give some information about whether a wetland performs a function, but no information is generated about levels of performance. The Washington State Wetland Rating System is a characterization based on five categories (sensitivity to disturbance, rarity, importance, ability to replicate, and relative level of functioning) (Ecology 1991).

Rating—classification based on a grade. Ratings usually group wetlands using the qualitative grades of high, medium, or low on a variety of scales such as the performance of a function or its value. The wetland evaluation technique or WET (Adamus et al. 1987) is probably the most widely used rating method.

Assessment—an estimate or determination of importance or value. This is the first level at which numbers are generated to represent an estimate of performance, value, or functional value of a function. All commonly used "rapid" numeric methods fall into this category. These methods only provide an assessment that is relative to some predetermined standard. They do not provide an assessment of actual levels of performance or value. The term "assessment" is one of the most commonly misused words in the lexicon of wetland scientists. Almost any method developed is now called an assessment, regardless of whether it might actually be a categorization, a rating, or a true assessment.

Evaluation—a determination or fixing of value. The fixing of value for any item is based on having a generally acceptable currency. Up to now the only currency used has been monetary, and evaluations of wetland functions have most often tried to generate dollar values based on different types of economic models such as the travel cost method, random utility model, hedonic techniques, contingent valuation method (Titre and Henderson 1989, Lipton et al. 1995), or willingness-to-pay method (Farber and Costanza 1987).