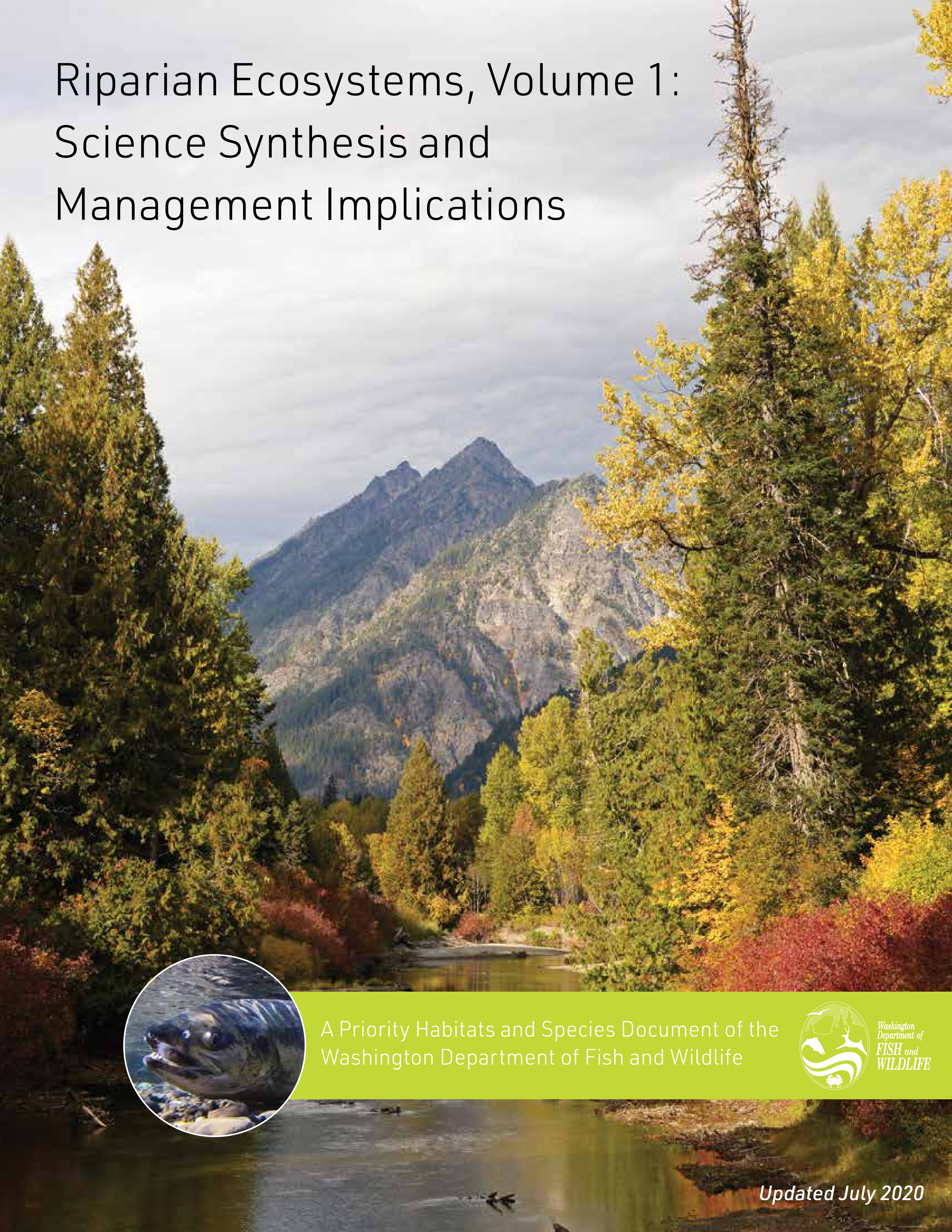
Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications



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Washington Department of Fish and Wildlife

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Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications

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Technical Editors
Timothy Quinn
George F. Wilhere
Kirk L. Krueger

Production Manager
Alison Hart

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Nichole Embertson, PhD	Whatcom Conservation District
Martin Fox, PhD	Muckleshoot Tribe
Aimee Fullerton, PhD	NOAA Fisheries
Marc Hayes, PhD	WA Department of Fish and Wildlife
Tim Hyatt	Skagit River System Cooperative
Jack Janisch	WA Department of Ecology
Peter Leinenbach	US Environmental Protection Agency
Chris May, PhD	Kitsap County
Michael Pollock, PhD	NOAA Fisheries
Jeremy Sikes	WA Department of Ecology
Carol Smith, PhD	WA State Conservation Commission
Steve Wondzell, PhD	US Forest Service

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While we acknowledge and appreciate all review comments, WDFW bears sole responsibility for the content of this document.

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Leopard Frog Lithobates pipiens/*Scott Fitkin, WDFW*

Preface

This Priority Habitats and Species (PHS) document of the Washington Department of Fish and Wildlife (WDFW) is provided in support of the agency's mission to preserve, protect, and perpetuate the state's fish, wildlife, and ecosystems while providing sustainable fish and wildlife recreational and commercial opportunities. WDFW works cooperatively with land use decision makers and landowners to facilitate solutions that accommodate local needs and needs of fish and wildlife. One of WDFW's important roles in land use decision making is that of technical advisor: we provide information about the habitat needs of fish and wildlife and the likely implications of various land use decisions for fish and wildlife.

The nine chapters of Volume 1 are a partial update of a 1997 document entitled *Management Recommendations for Washington's Priority Habitats: Riparian.* This document, called *Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications* is a partial update because it addresses only aquatic species. Riparian needs of terrestrial species will be updated in the future. Until that update is complete, readers can consult the 1997 document, available at **http://wdfw.wa.gov/publications/00029/**, for information about riparian ecosystems and terrestrial species.

Priority Habitats are places that warrant special consideration for protection when land use decisions are made. For more information, see *Washington Department of Fish and Wildlife. 2008. Priority Habitat and Species List,* available at **https://wdfw.wa.gov/species-habitats/at-risk/phs.**

The PHS program provides land use decision support to clients such as local governments, tribes, government agencies, non-governmental organizations, and landowners. PHS consists of a PHS List, PHS Maps available at **https://wdfw.wa.gov/species-habitats/at-risk/phs/maps**, PHS Management Recommendations, and technical assistance from our regional habitat biologists.

This PHS document compliments a family of PHS documents including *Landscape Planning for Washington's Wildlife: Managing for Biodiversity in Developing Areas* and *Land Use Planning for Salmon, Steelhead and Trout: A land use planner's guide to salmonid habitat protection and recovery,* available at **https://wdfw.wa.gov/species-habitats/at-risk/phs/recommendations**.

List of Acronyms

BAS	best available science
BFW	bankfull width
CMZ	channel migration zone
DBH	diameter at breast height
DNR	(Washington) Department of Natural Resources
EPA or (US)EPA	United States Environmental Protection Agency
FEMAT	Forest Ecosystem Management Assessment Team
HCP	habitat conservation plan
IMW	intensively monitored watersheds
LWD	large woody debris
NOAA	National Oceanographic and Atmospheric Administration
NRC	National Research Council
PHS	Priority Habitats and Species
PNW	Pacific Northwest
RCW	Revised Code of Washington
RMZ	riparian management zone
SMA	Shoreline Management Act
SPTH	site-potential tree height
SPTH_{200}	site-potential tree height at tree age equal to 200 years
TAG	Technical Advisory Group
USFS	United States Forest Service
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
WAC	Washington Administrative Code
WDFW	Washington Department of Fish and Wildlife
WNHP	Washington Natural Heritage Program
WSAS	Washington State Academy of Sciences

Bank of the Chehalis River/Kaysie Cox, WDFW

Chapter 1. Introduction

By: Timothy Quinn, Kirk L. Krueger, and George F. Wilhere

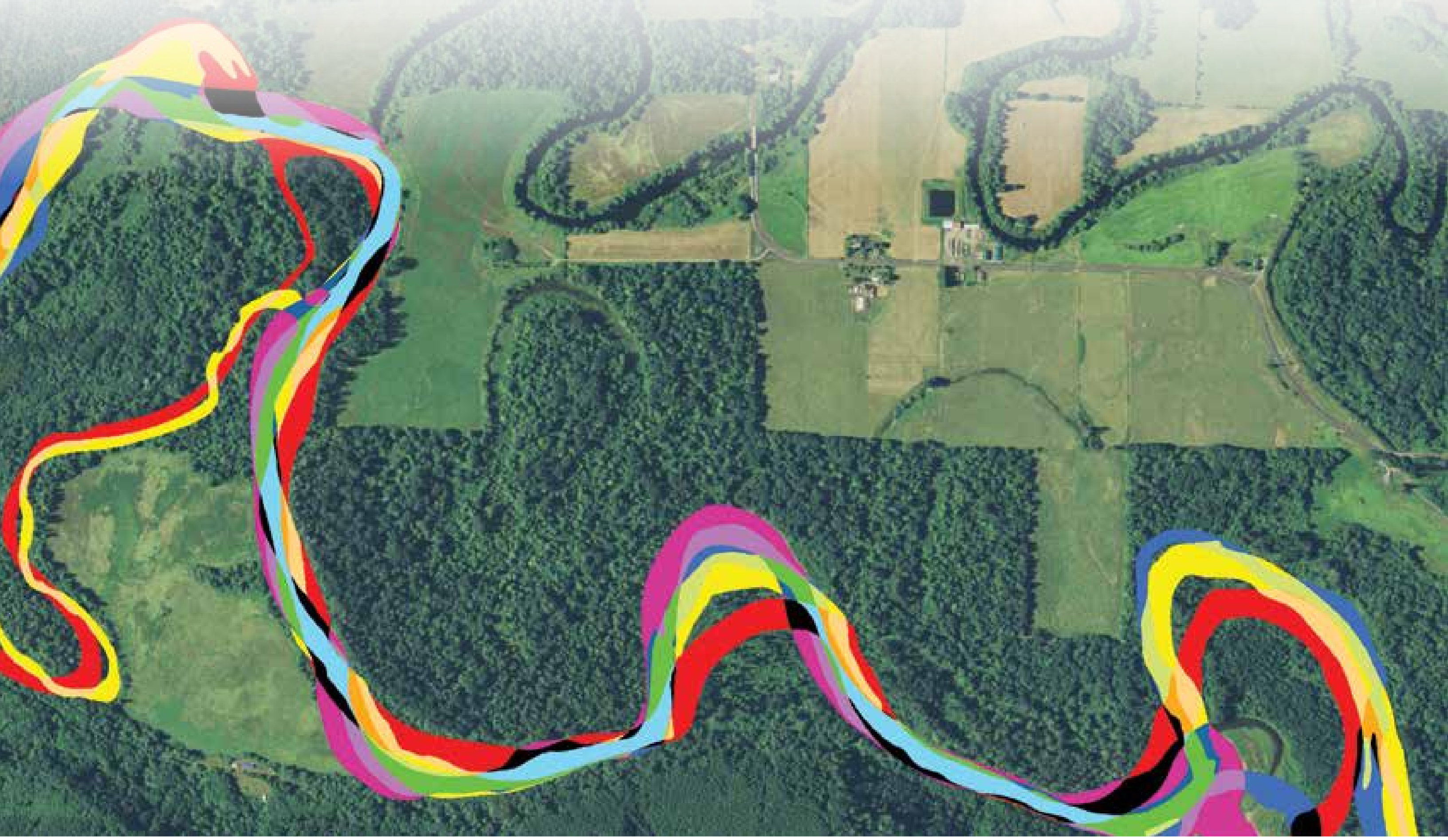
1.1. Document Description

Volume 1 is part one of a two-volume set. It contains reviews and syntheses of scientific literature for the purpose of informing the development of policies related to management of riparian areas and watersheds of Washington State. Volume 1 adds additional information to the science summarized in the PHS report titled *Management Recommendations for Washington's Priority Habitats: Riparian* (Knutson and Naef 1997).

Volume 1 was designed to answer the following three general questions:

- What is currently known about the key ecological functions of riparian areas?
- How do riparian areas and watersheds affect the freshwater habitats of fish and wildlife?
- How do human activities affect the capacity of riparian areas and watersheds to provide habitat for fish and wildlife in rivers and streams?

Volume 1 is intended to serve two goals. The first goal is to be a source of best available science (BAS) for understanding how riparian areas and surrounding



Chehalis River meander paths between 1938 and 2013/Ken Pierce, WDFW watersheds affect ecological functions and aquatic habitats. The second goal is to provide a scientific foundation for management recommendations presented in Volume 2, which are consistent with Washington Department of Fish and Wildlife's (WDFW) mandate (RCW 77.04.012) to protect, preserve, and perpetuate Washington's fish and wildlife. Both goals also relate to WDFW's responsibility to support and provide technical assistance for conservation and protection measures necessary to preserve or enhance anadromous fisheries under the Growth Management Act (GMA; RCW 36.70A.172) and the Shoreline Management Act (SMA; WAC 173-26-221(5)(b)).

Volume 1 does not directly address instream flows as affected by water withdrawal for domestic and commercial uses, or instream water quality, both of which are regulated by the Washington Department of Ecology. However, because water quality is affected by land use, we discuss this issue in the context of riparian and watershed science. In addition, Volume 1 does not specifically address riverine wetlands, i.e., wetland associated with rivers and streams and commonly found in floodplains. Rather, riverine wetlands are usually considered part of riparian ecosystems, which are the focus of this document. Other specific wetland management guidance is available from Washington Department of Ecology (**https://ecology.wa.gov/Water-Shorelines/Wetlands**).

Volume 2 translates these science reviews and syntheses into land use guidance for local governments and other organizations to conserve watershed processes and riparian ecosystems in support of aquatic species and their habitats. The guidance presented in Volume 2 is not in and of itself "best available science." Rather, it represents the recommendations of WDFW as to how a local government could include the best available science in policies, plans, and regulations to protect riparian ecosystems and their associated aquatic habitats.

WDFW recognizes that natural resource management and land use policies must often balance competing ecological, economic, and social goals (Wilhere and Quinn 2018). In fact, one of WDFW's conservation principles is that management decisions should integrate ecological, economic, and social perspectives. This volume addresses only the ecology of riparian areas within the context of watershed conditions; it does not cover economic and social aspects. Determining an acceptable balance requires a political process involving stakeholders, local governments, tribes, and state agencies. Such a process would establish society's goals for freshwater habitats for salmon and other species of fish and wildlife, and lead to management for achieving those goals. This volume can help to inform that process.

1.1.1. Ecosystems

The ecosystem, a fundamental concept in natural resource management, can be defined as "a spatially explicit unit of the Earth that includes all of the organisms, along with all components of the abiotic environment" (Likens 1992). Throughout this document we use the term ecosystem composition to mean "the different parts (components) which something is made of" (Oxford Dictionary 2015). We define ecosystem structure as the "arrangement of and relations among the parts or elements of something complex" (Oxford Dictionary 2015). We define ecosystem function(ing) to include the first and second definitions of ecological function by Jax (2005): 1) the process, or the cause-effect-relationship underlying two or more interacting components, and 2) the sum of processes that sustain the system. Note that the definitions of functions we use here are broader and more inclusive than "functions" defined solely as ecosystem outputs that benefit humans, namely goods and services (Jax 2005; MEA 2005).

In this volume, we explicitly consider the composition, structure, and function of riparian ecosystems in the context of the surrounding watershed. This is especially important in light of the challenges of managing ecosystems sustainably and the risks associated with scientific uncertainty, i.e., the consequences of being wrong.

The composition, structure, and function of ecosystems describe the organization of biotic and abiotic elements and the processes occurring among elements. These concepts are closely related but not equally well studied nor understood. For example, describing an ecosystem's composition and structure is easier than describing its functions (Hunter et al. 1988; Noss 1992). Likewise, management objectives are often expressed in terms of composition and structure, not functions, because they are easier to characterize.

Composition and structure versus function have a chicken-and-egg relationship. That is, functions (processes) are in large part responsible for maintaining composition and structure, but some functions could not exist without an ecosystem's particular composition and structure. Myster (2001), for instance, argues that ecosystem structure can be defined as the minimal organization necessary for a function to operate. Others argue that managing for functions (Beechie and Bolton 1999) is likely to be more successful than managing for composition and structure because processes likely exert stronger influences on composition and structure than the other way around (Odum 1953), and processes are more directly affected by land use activities (Reid 1998). Furthermore, some land use laws or regulations stipulate no net loss of ecological functions, not no net loss of composition and structure (RCW 90.58, WAC 173-26-186, WAC 365-196-830), however, compliance is often determined through evaluating ecosystem composition and structure, not functions. Such conceptual ambiguity suggests that we consider the entire ecosystem, i.e., composition, structure, and functions, when studying riparian areas, and furthermore, that land managers understand how composition and structure manifest ecological functions. The following chapters synthesize our review of the relationships among composition, structure, and functions of riparian ecosystems in the Pacific Northwest.

1.1.2. Riparian Ecosystems

WDFW has adopted the following conceptual definition of riparian ecosystems:

Riparian areas are transitional between terrestrial and aquatic ecosystems and are distinguished by gradients in biophysical conditions, ecological processes, and biota. They are areas through which surface and subsurface hydrology connect waterbodies with their adjacent uplands. They include those portions of terrestrial ecosystems that significantly influence exchanges of energy and matter with aquatic ecosystems (i.e., a zone of influence).

Riparian ecosystem/Scott Fitkin, WDFW

Riparian ecosystems are sometimes referred to by different names: riparian areas, riparian zones, or riparian habitats. To compound confusion, the "riparian zone" is sometimes identified as a distinctive area within riparian ecosystems (Sedell et al. 1989; Naiman et al. 1992; Steiner et al. 1994). Some definitions include the adjacent waters whereas others do not. Unless stated otherwise, we consider riparian area, riparian habitat, riparian corridor, and riparian ecosystem to be synonyms. Furthermore, our definition of riparian ecosystem does not include adjacent waters (i.e., rivers or streams) and recognizes the riparian zone as a distinctive area within riparian ecosystems. The riparian zone contains wet or moist soils and plants adapted to growing conditions associated with periodically saturated soils.

Recent definitions reflect common conceptual understanding and identify similar composition, structure, and functions, including distinct plant communities and soil types; disturbance regimes unique to fluvial systems; gradients in soil, vegetation, and microclimate that manifest an ecotone between terrestrial and aquatic environments; and essential interactions between terrestrial and aquatic ecosystems. Riparian areas are the loci of terrestrial and aquatic interactions, however, the exact boundaries of riparian ecosystems can be difficult to delineate because: 1) ecosystems are not uniquely identified entities (Karr 1996); 2) the environmental heterogeneity of riparian areas is expressed in a variety of plant life history strategies and successional patterns; and 3) some ecosystem functions depend on environmental setting (Naiman et al. 1998).

Following Naiman et al. (1998), our definition of riparian ecosystem includes the active floodplain, including riverine wetlands, and the terraces or adjacent uplands that directly contribute organic matter or large wood to the active channel or active floodplain (Figure 1.1 and 1.2). Terraces and adjacent uplands that contribute organic matter or large wood are located in the "zone of influence." The active channel is defined by the lower limit of continuous riparian vegetation (Naiman et al. 1998), and it may be delineated by absence of both moss on rocks and rooted vegetation (USFS 2008). The upper most elevation of the active channel is sometimes equated with the ordinary high water mark. The active floodplain is located between the active channel and adjacent terrace or hillslopes (Fetherston et al. 1995; Harris 1987). Depending on the watershed, the flood return interval of the active floodplain varies between 1 and 10 years (Wolman and Leopold 1957; Ward and Stanford 1995; Lichvar et al. 2006; Williams et al. 2009; BLM 2015).

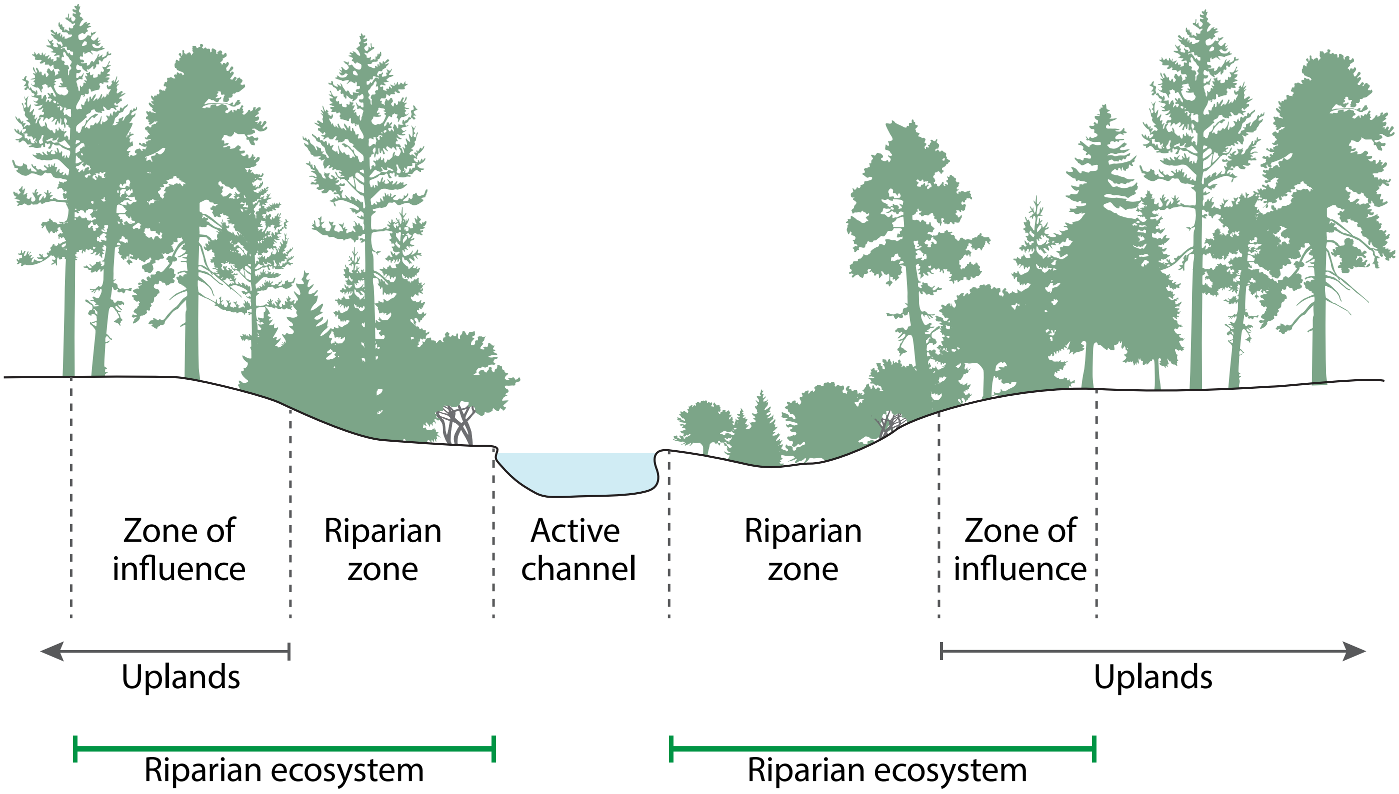


Figure 1.1. A generalized diagram of the riparian ecosystem. The NRC (2002) states that, "Riparian areas are transitional between terrestrial and aquatic ecosystems and are distinguished by gradients in biophysical conditions, ecological processes, and biota. They are areas through which surface and subsurface hydrology connect waterbodies with their adjacent uplands. They include those portions of terrestrial ecosystems that significantly influence exchanges of energy and matter with aquatic ecosystems (i.e., the zone of influence)." The portion of the ecosystem characterized by moist soils and plants adapted to periodically saturated soils is the riparian zone. The width of the riparian ecosystem is typically based on riparian functions (i.e., wood recruitment to the stream in forested regions and the pollution removal function in dryland regions).

Riparian ecosystems are priority habitats in part because wildlife occurs more often and in greater variety in riparian areas than in any other habitat type. "Natural riparian corridors are the most diverse, dynamic, and complex biophysical habitats on the terrestrial portion of the earth" (Naiman et al. 1993). Although riparian ecosystems constitute a small portion of the surface landscape, approximately 85% of Washington's wildlife species are known to use riparian areas associated with rivers and streams (Knutson and Naef 1997). Of these, 170 species including 134 mollusks, 11 amphibians, 3 reptiles, 10 birds, and 9 mammals may be riparian obligates, i.e., require riparian habitat to survive (T. Quinn, unpublished). In addition, habitat for many upland and aquatic species is directly enhanced by the presence of adjacent riparian areas. A principal reason for high fish and wildlife diversity is that riparian ecosystems are exceptionally productive. Riparian

Western Toad Anaxyrus boreas *egg strings/Scott Fitkin, WDFW* ecosystems are characterized by the availability of water, milder microclimate, and relatively fertile soils. These factors enhance the productivity of plant communities and support a complex food web that includes a rich variety and abundance of fish, invertebrates, amphibians, plants, bacteria, fungi, reptiles, birds, and mammals (Cummins 1974; Johnson and Carothers 1982). It is not surprising then, that $1.4 billion (57%) of $2.5 billion spent to recover salmon in Pacific Northwest since 2000 has been directed at enhancing, restoring, and maintaining salmon habitat (e.g., large wood supplementation, riparian planting, fine sediment remediation, land acquisition) previously provided in large part by functioning riparian areas (NOAA 2015).

In addition to their essential role as fish and wildlife habitats, riparian areas provide other significant benefits to humans (Naiman and Bilby 1998; NRC 2002). Briefly, riparian areas help provide a variety of ecosystem goods and services: provisioning services such as food and water; regulating services such as decreasing flood flows; supporting services such as nutrient cycling, sediment and pollutant filtering, and carbon sequestration; and cultural services such as recreational, spiritual, and other nonmaterial benefits (MEA 2005). These services provide real but often unquantified economic benefits to individuals and society; benefits that largely go unnoticed until they are lacking. According to the NRC (2002), protection and restoration of riparian areas should be a national goal because they have a major influence on achieving important national standards of the Clean Water Act, the Endangered Species Act, and flood damage control programs.

This document includes consideration of river-associated wetlands but does not consider other types of wetlands, which are covered by the Washington

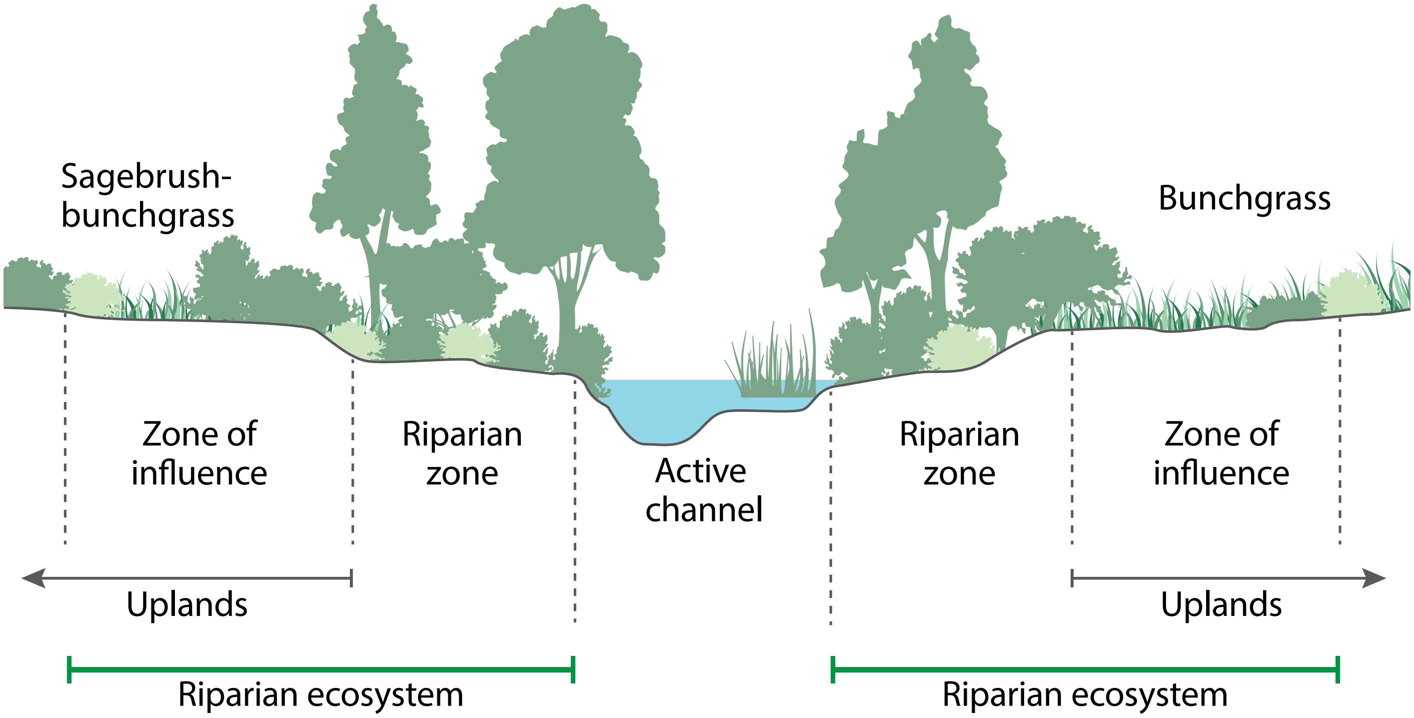


Figure 1.2. Dryland Riparian Ecosystem. The riparian ecosystem consists of two zones: riparian and zone of influence. The riparian zone extends from the edge of the active channel towards the uplands. This zone includes areas where terrestrial biota and soils are influenced, at least periodically, by surface or subsurface waters. Beyond this is the riparian "zone of influence." This includes areas where ecological processes significantly influence the stream (diagram modified from USFS 2004).

Department of Ecology, or areas bordering marine or estuarine (brackish) waters despite the potential applicability of this work to those environmental settings. A 2009 review of riparian functions of marine shorelines is available at **https://wdfw.wa.gov/publications/00693**.

1.2. Scope of Volume 1

1.2.1. Riparian Ecosystem Management

Riparian ecosystem management is often couched in terms of maintaining riparian functions thought to be important to fish and wildlife and their habitats (e.g., Bolton and Shellberg 2001, Everest and Reeves 2007). Some commonly identified riparian functions are described in FEMAT (1993) and the Forests and Fish Report (DNR 1999). The riparian ecosystem functions we discuss in this document, including some associated compositional and structural elements, are organized by chapter:

- Chapter 2. *Stream Morphology*—the interplay of water, sediment, and vegetation on channel form and riparian areas
- Chapter 3. *Wood*—effects of wood on channel morphology and habitat, and the recruitment of wood from riparian areas
- Chapter 4. *Stream Temperature*—effects of hydrology and shading on the thermal regime of streams
- Chapter 5. *Pollutant Removal*—the interception or filtration of fine sediments, excessive nutrients, pathogens, pesticides, and other contaminants in overland and shallow subsurface flows
- Chapter 6. *Nutrients Dynamics in Riparian Ecosystems*—dynamics and fate of the primary macro-nutrients: nitrogen, phosphorus, and carbon
- Chapter 7. *Riparian Areas of the Columbia Plateau*
- Chapter 8. *Watersheds*—relationship of watershed-scale management to the goal of achieving beneficial effects for fish and wildlife

Riparian functions are affected and in turn affect aquatic and terrestrial systems at multiple spatial scales. We focus much attention on riparian functions at the site (e.g., stream reach or land parcel) scale for three reasons. First, riparian areas are disproportionately important, relative to area, for fish and wildlife and other ecosystem services (Naiman and Bilby 1998; NRC 2002). Second, local governments and individual landowners, important users of WDFW's PHS guidance, typically manage at the parcel scale. Most land use activity, outside of public lands, occurs at the site-scale on ownership parcels that are relatively small compared to the watershed in which they occur. Third, effective and efficient conservation of fish and wildlife habitats requires management at multiple scales and thus good site-scale riparian management is an essential complement to good watershed management (Chapter 8), although approaches at different scales can vary dramatically.

1.2.2. Watershed Management

Even when all riparian functions are protected, rivers, streams, and riparian ecosystems are not immune to the effects of upland management. To varying degrees, riparian and stream structure, composition, and function reflect land uses and conditions throughout watersheds.

Regardless of its size, no ecosystem is entirely closed. Upslope activities can considerably alter the magnitude and timing of stream flows; the mobilization, routing, and storage of sediment, wood, and fine organic matter; and the quality of water, thereby influencing riparian ecosystem functions (NRC 2002; Figure 1.3). Thus, the management of riparian ecosystems at the site or reach scale should be considered just one important part, not the sole requirement, of holistic watershed management for the protection of fish and wildlife and other values. We include Chapter 8, which focuses on watershed-scale issues that are important to the management of aquatic systems as a whole, with the intent of increasing the effectiveness of collective riparian conservation efforts. This chapter provides a conceptual framework that identifies the key watershed processes and their primary influences by, on, and through riparian areas and adjacent watercourses.

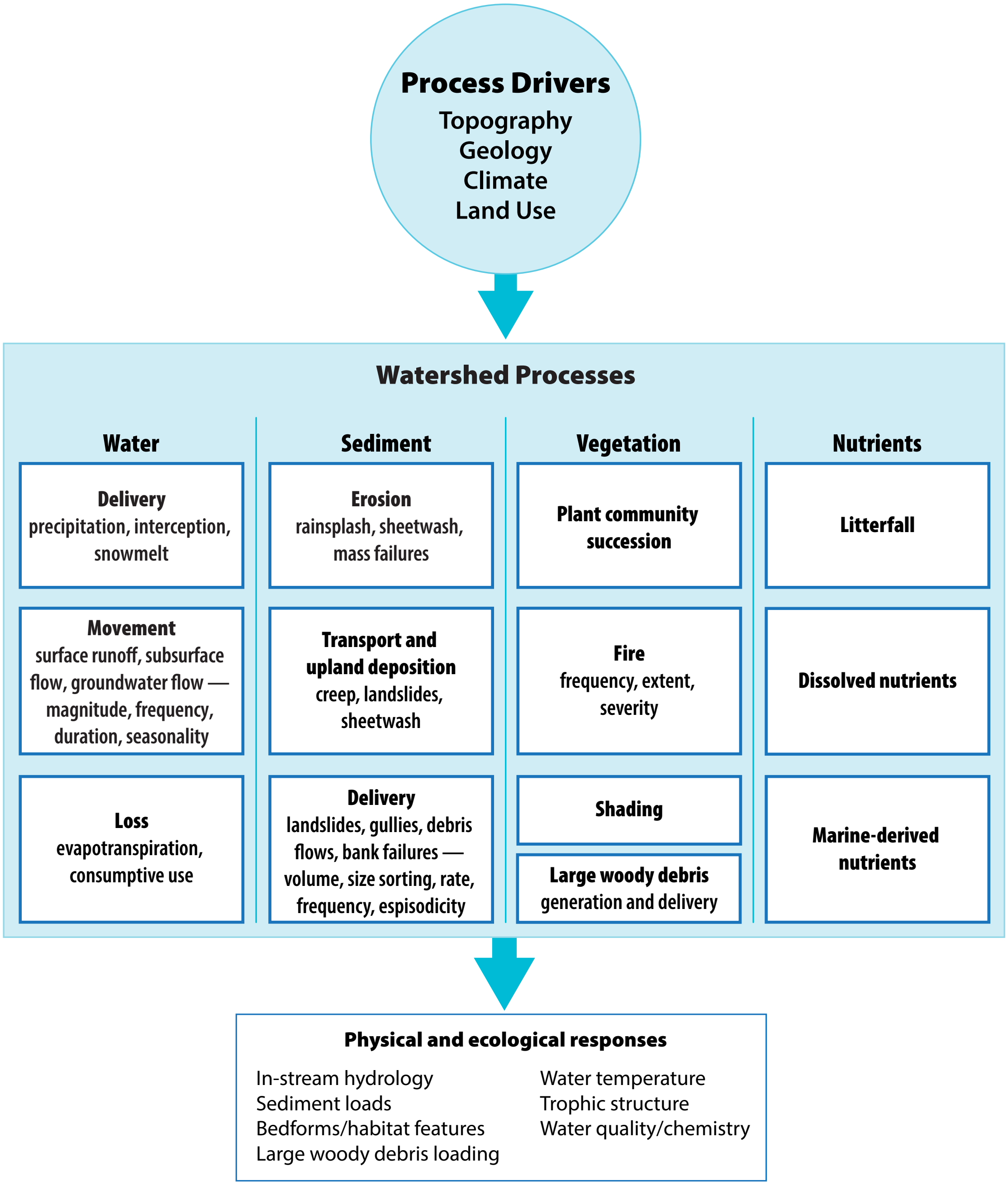


Figure 1.3. Conceptual framework for the hierarchical relationship of invariant, large-scale "process drivers," the suite of watershed processes that are determined by these drivers, and the instream physical and biological responses to those processes. Multiple additional interactions between elements and levels are not shown on this diagram to emphasize the primary influences; but in any given setting, one or more of these secondary interactions may temporarily achieve equivalent importance. This framework embraces the definition of watershed processes from Stanley et al. (2011): "[t]he dynamic physical and chemical interactions that form and maintain the landscape and ecosystems."

Devil's Club Oplopanax horridus *is common in wet soils of western Washington riparian areas/Ned Pittman, WDFW*

1.2.3. Science Synthesis

Chapter 9, *Science Synthesis to Management Implications,* of this volume summarizes important scientific finding from chapters 1-8 and provides a brief discussion of scientific themes that can inform thoughtful approaches to protecting aquatic systems.

1.2.4. Scope: Protection and Restoration

Protection and restoration of riparian ecosystems and watersheds are both essential for Washington's fish and wildlife, and especially for endangered and threatened salmonids. However, planning, management practices, regulations, and legal context for protection are different from those for restoration, and thoroughly covering both topics was beyond our scope. Thus, restoration is not explicitly incorporated in the scope of this document. Nevertheless, much of the science presented in this document can inform restoration activities. For instance, while this document does not review scientific literature on instream large wood placement, a common restoration activity, it does summarize large wood recruitment and reference conditions for the amount and size of instream large wood. Thus, the body of science that informs protection also helps to inform restoration, at least in terms of establishing resource goals and objectives.

1.3. Document Development

We employed the following approach to developing this document. We first established a Technical Advisory Group (TAG) of scientists from the region. Second, with guidance from the TAG, we conducted a literature review designed to inform the general state of knowledge regarding ecosystem protection and another literature review more focused on recent developments in riparian ecology and management. Third, we contracted with regional topic experts to author and co-author chapters. WDFW staff provided conceptual and technical guidance as possible. Finally, we subjected the draft final documents to several rounds of peer review. Each of these steps is discussed below.

1.3.1. Technical Advisory Group

The TAG helped provide overall science guidance, that is, contribute expertise and scientific perspective, provide important references and other sources of information, and review draft products. The TAG was composed of 13 individuals who were selected by virtue of their scientific expertise as opposed to affiliation with particular organization or stakeholder group. However, given the broad potential applicability of this document, it is conceivable that TAG members belong to one or more stakeholder groups. All TAG members were expected to attend three all-day meetings over the course of the project. Members of the TAG volunteered their time, except one who was paid a small stipend because she was employed by a non-profit organization. The first meeting was designed to introduce TAG members to the nature of the PHS update, to discuss different approaches to conducting a literature review and synthesizing new scientific information, and to secure their ongoing commitment to helping WDFW meet high scientific standards. The second and third TAG meetings were designed to solicit review and feedback from TAG members on draft products. As part of their review process, we invited all TAG members to edit draft chapters and provide additional citations for WDFW's considerations. Although the TAG played a key role in advising WDFW on the science of riparian management and conservation, WDFW bears sole responsibility for the contents of this document.

1.3.2. Literature Review

We organized summaries around ecological functions including stream channel morphology, wood, pollution removal, nutrients, water temperatures, and watershed processes. Note that many of the issues we address here are directly or indirectly related to fish and wildlife habitats and other ecosystem services. We did not, however, explicitly consider cultural services that are important values for many stakeholders (e.g., spiritual, cultural and aesthetic values of riparian areas).

We enlisted the assistance of scientists with specific areas of expertise as technical advisors, authors, and informal and formal reviewers to help ensure that our summaries of pertinent scientific information were sufficiently accurate and inclusive. We were most interested in characterizing the state of knowledge and important, recent findings about each topic. That is, we tried to assesses what we know, how well we know it, and what we do not know. The summary statements in each chapter are important to the development of management recommendations because they often identify important management challenges that are not well described in the published literature (e.g., the importance of habitat connectivity to fish population persistence). To this end, when appropriate, we encouraged authors to consider the broad body of scientific information and how it might be extrapolated to unstudied locales; in other words, to consider the body of literature with regard to our ability to make reliable predictions.

Where information existed and as time allowed, we also attempted to summarize how riparian composition, structure, and functions differ among natural plant communities. In most cases, we used the broadest division called Major Vegetational Areas, which is made up of two groups in Washington State, i.e., Forested Regions and Steppe Regions (Franklin and Dyrness 1988). Regions can be further subdivided into forested zones (e.g., *Tsuga heterophylla* Zone) and steppe communities (e.g., shrub-steppe with *Artemisia tridentata*) that contain similar climax plant communities influenced by similar disturbance regimes. When information was available, for example, in the case of large wood in forested regions, we report that information at the finer division within the vegetation classification.

In addition, and as time and resources allowed, we summarized some potential effects of common land use activities on stream and riparian areas as a precursor to providing useful land use advice. We considered four general categories of land use: 1) urban, which includes commercial, industrial, urban, suburban, and rural development, 2) agricultural, which includes cultivated crops and grazing uses, 3) forestry, which includes timber management and associated activities such as

Chapter 9 summarizes scientific findings from chapters 1 through 8 and includes a brief discussion of scientific themes that can inform thoughtful approaches to protecting aquatic systems.

road building and maintenance, and 4) other activities (e.g., mining), which were not discussed due to the paucity of available research results.

We started this work by assembling topical literature reviews including Bolton and Shellberg (2001), Pizzimenti (2002), Bezener and Bishop (2005), Mayer et al. (2005), and Naiman et al. (2005). We compiled relevant literature cited in these review articles, from other reference materials we encountered, and from recommendations of TAG members. We also did an extensive literature search on computerized databases. We mostly focused our literature search on peer reviewed articles published after 1993, published in English, and conducted in proximity to Washington State. This last criterion was designed to focus on studies thought most relevant to Washington State's ecological conditions.

The North Fork Skokomish River runs through mature conifer forests of the Olympic National Park/Tim Quinn, WDFW

We also documented the type of review the study received according to RCW 34.05.271 (Appendix 1). Studies with no record of review were not considered further. As we (authors) reviewed the published literature, we noted important study findings that we believed would be especially useful for writing the PHS document. For empirical studies, our notes typically included the primary research questions or objectives, key results, management implications, and study limitations or assumptions. For literature reviews and syntheses, our notes summarized main conclusions by subject area. The intent here was to capture a sufficient level of detail for each paper so that we could use our collective notes as sources for writing the synthesis.

We found a massive quantity of literature related to riparian and watershed science despite focusing on information published after 1993. Due to the sheer volume of information, we could not cover all the literature in depth, so we covered those topics that were most relevant for management of riparian areas in Washington State. This document provides a comprehensive review on the current state of the science regarding certain riparian functions. Further, as described above, we enlisted the services of experts with specific areas of expertise as technical advisors, authors, and informal and formal reviewers to help ensure that our summaries of pertinent scientific information were sufficiently accurate and inclusive.

1.3.3. Peer Review

The peer review for Volume 1 consisted of a series of iterative steps, wherein we tried to gain as much agreement about the science synthesis as possible among those involved in the writing and reviewing process. Our intent was to provide the best available science. As described above, the TAG reviewed and often edited incomplete chapter drafts during the literature review and synthesis stage of the project. These draft syntheses were then given to topic experts to complete.

In cases, where WDFW lacked sufficient topic expertise, we provided stipends up to $10,000 to outside authors (i.e., contractors) for completing a chapter. The steps involved in completion of drafts included adding one or more conceptual models describing the system and its functioning and a synthesis or summary of the important findings over the last approximately 20 years. These steps often entailed a series of editing and reediting by WDFW staff and contractors. Once completed by contractors and returned to WDFW, the drafts were reviewed a final time by TAG members. This last review by TAG members was to ensure that completion of the chapters was consistent with their earlier guidance and with their understanding of the literature. WDFW attempted to resolve all disagreement that occurred between contractors and TAG members and provided a written description of how those issues were resolved to both the contractor and the TAG. After final WDFW review, the document was submitted for review by the Washington State Academy of Sciences (WSAS).

We enlisted the WSAS to conduct the final, independent peer review of the PHS document. Modeled after the National Academy of Science, which is charged with providing independent, objective advice to the nation on matters related to science and technology, the WSAS provides expert scientific analysis to inform public policy making at the state level (WSAS 2015). The final peer review with the WSAS was also iterative. That is, WDFW in consultation with authors and TAG members, addressed WSAS comments, made appropriate changes to the document, and then discussed those comments with the WSAS in an attempt to resolve differences. This process proceeded until we reached agreement about scientific contents and conclusions.

1.4. Literature Cited[1]

Beechie, T., and S. Bolton. 1999. An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds. Fisheries 24:6-15. (i)

Bezener, A.M., and C.A. Bishop. 2005. Literature review of riparian habitat requirements for aquatic and terrestrial wildlife and its application to habitat restoration projects: a case example in the South Okanagan and Similkameen Valleys, British Columbia. Technical Report Series No. 379. Canadian Wildlife Service, Pacific and Yukon Region, British Columbia. (viii)

BLM (Bureau of Land Management). 2015. Riparian area management: proper functioning condition assessment for lotic areas. Technical Reference 1737-15. National Operations Center, Denver, Colorado. (viii)

Bolton, S.M., and J. Shellberg. 2001. Ecological issues in floodplains and riparian corridors. Center for Streamside Studies, University of Washington, Seattle. (viii)

Cummins, K.W. 1974. Structure and function of streams ecosystems. BioScience 24:631-641. (i)

DNR (Washington Department of Natural Resources). 1999. Forests and Fish Report. Forest Practices Division, Olympia. (viii)

Everest, F.H., and G.H. Reeves. 2007. Riparian and aquatic habitats of the Pacific Northwest and southeast Alaska: ecology, management history, and potential management strategies. PNW-GTR-692. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management: an ecological, economic and social assessment. U.S. Department of Agriculture and U.S. Department of the Interior, Portland, Oregon. (viii)

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[1] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Fetherston, K.L., R.J. Naiman, and R.E. Bilby. 1995. Large woody debris, physical process, and riparian forest succession in montane river networks of the Pacific Northwest. Geomorphology 13:133-144. (i)

Franklin, J.F., and C.T. Dyrness. 1988. Natural vegetation of Oregon and Washington. Oregon State University Press, Corvallis. (i)

Harris, R.R. 1987. Occurrence of vegetation on geomorphic surfaces in the active floodplain of a California alluvial stream. American Midland Naturalist 118:393-405. (i)

Hunter, M.L. Jr., G.L. Jacobson Jr., and T. Web III. 1988. Paleoecology and the coarse filter approach to maintaining biological diversity. Conservation Biology 2:375-385. (i)

Jax, K. 2005. Function and "functioning" in ecology: what does it mean? Oikos 111:641-648. (i)

Johnson, R.R., and S.W. Carothers. 1982. Riparian habitat and recreation: interrelationships and impacts in the southwest and Rocky Mountains region. Eisenhower Consortium Bulletin 12. Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado. (viii)

Karr, J.R. 1996. Ecological integrity and ecological health are not the same. Pages 97-109 *in* P.C. Schulze, R. Frosch, and P. Risser, editors. Engineering within ecological constraints. National Academy of Engineering, Washington, D.C. (viii)

Knutson, K.L., and V.L. Naef. 1997. Management recommendations for Washington's priority habitats: Riparian. Washington Department of Fish and Wildlife, Olympia. (iv)

Lichvar, R.W., D.C. Finnegan, M.P. Ericsson, and W. Ochs. 2006. Distribution of ordinary high water mark (OHWM) indicators and their reliability in identifying the limits of "waters of the United States" in arid southwestern channels. ERDC/CRREL TR-06-5. U.S. Army Corps of Engineers, Washington, D.C. (viii)

Likens, G. 1992. An ecosystem approach: its use and abuse. Ecology Institute, Oldendorf/Luhe Germany. (i)

Mayer, P.M., S.K. Reynolds, M.D. McMutchen, and T.J. Canfield. 2005. Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: a review of current science and regulations. EPA/600/R-05/118. USEPA, Office of Research and Development, Washington, D.C. (viii)

MEA (Millennium Ecosystem Assessment). 2005. Ecosystems and human well-being: synthesis. Island Press, Washington, D.C. (i)

Myster, R.W. 2001. What is ecosystem structure? Caribbean Journal of Science 37(1-2):132-134. (i)

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. O'Conner, P.L. Olsen, and E.A. Steel. 1992. Fundamental elements of ecologically healthy watersheds in the Pacific Northwest Coastal Ecoregion. Pages 127-188 *in* R.J. Naiman, editor. Watershed management: balancing sustainability and environmental change. Springer, New York. (i)

Naiman R.J., and R.E. Bilby, editors. 1998. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer Science & Business Media, New York. (i)

Naiman, R.J., H. Decamps, and M.E. McClain. 2005. Riparia: ecology, conservation, and management of streamside communities. Elsevier Academic Press, San Francisco. (i)

Naiman, R.J., H. Decamps, and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3:209-212. (i)

Naiman, R.J., K.L. Fetherston, S.J. McKay, and J. Chen. 1998. Riparian forests. Pages 289-323 *in* R.J. Naiman and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer, New York. (i)

NOAA (National Oceanic and Atmospheric Administration). 2015. Pacific Coastal salmon recovery funds: project and performance metric database. Available: **https://www.webapps.nwfsc.noaa.gov/apex/f?p=309:13**. (January 2019). (viii)

Noss, R.F. 1992. The Wildlands Project: land conservation strategy. Wild Earth (Special Issue):10-25. (i)

NRC (National Research Council). 2002. Riparian areas: functions and strategies for management. National Academy Press, Washington, D.C. (i)

Odum, E.P. 1953. Fundamentals of ecology. W. B. Saunders, Philadelphia. (i)

Oxford Dictionary. 2015. Available: **https://www.lexico.com/en/definition/structure**. (January 2019). (viii)

Pizzimenti, J. 2002. Efficacy and economics of riparian buffers on agricultural lands, State of Washington. GEI Consultants, Inc., Englewood, Colorado. (viii)

Reid, L.M. 1998. Cumulative watershed effects and watershed analysis. Pages 476-501 *in* R.J Naiman and R. E. Bilby, editors. River ecology and management. Springer-Verlag, New York. (i)

Stanley, S., S. Grigsby, D.B. Booth, D. Hartley, R. Horner, T. Hruby, J. Thomas, P. Bissonnette, R. Fuerstenberg, J. Lee, P. Olson, and G. Wilhere. 2011. Puget Sound characterization. volume 1: the water resources assessments (water flow and water quality). Publication #11-06-016. Washington State Department of Ecology, Olympia. Available: **https://fortress.wa.gov/ecy/publications/documents/1106016.pdf**. (January 2019). (iv)

Steiner, F., S. Pieart, E. Cook, J. Rich, and V. Coltman. 1994. State wetlands and riparian area protection programs. Environmental Management 18:183–201. (i)

Sedell, J.R., F.H. Everest, and D.R. Gibbons. 1989. Streamside vegetation management for aquatic habitat. Pages 115-125 *in* Proceedings of the National Silviculture Workshop, U.S. Forest Service, Washington, D.C. (viii)

USFS (U.S. Forest Service). 2008. Stream simulation: an ecological approach to providing passage for aquatic organisms at road-stream crossings. National Technology and Development Program, San Dimas, California. (viii)

Ward, J.V., and J.A. Stanford. 1995. Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. Regulated Rivers: Research & Management 11:105-119. (i)

Williams, P.B., E. Andrews, J.J. Opperman, S. Bozkurt, and P.B. Moyle. 2009. Quantifying activated floodplains on a lowland regulated river: its application to floodplain restoration in the Sacramento Valley. San Francisco Estuary and Watershed Science 7(1). (i)

Wolman, M.G., and L.B. Leopold. 1957. River flood plains: some observations on their formation. Geological Survey Professional Paper 282-C. U.S. Government Printing Office, Washington, D.C. (i)

Wilhere, G.F., and T. Quinn. 2018. How wide is wide enough?: values, science, and law in riparian habitat conservation. Natural Resources Journal 58:279-318. (i)

WSAS (Washington State Academy of Science). 2015. Available: **http://www.washacad.org/aboutus/**. (January 2019). (v)

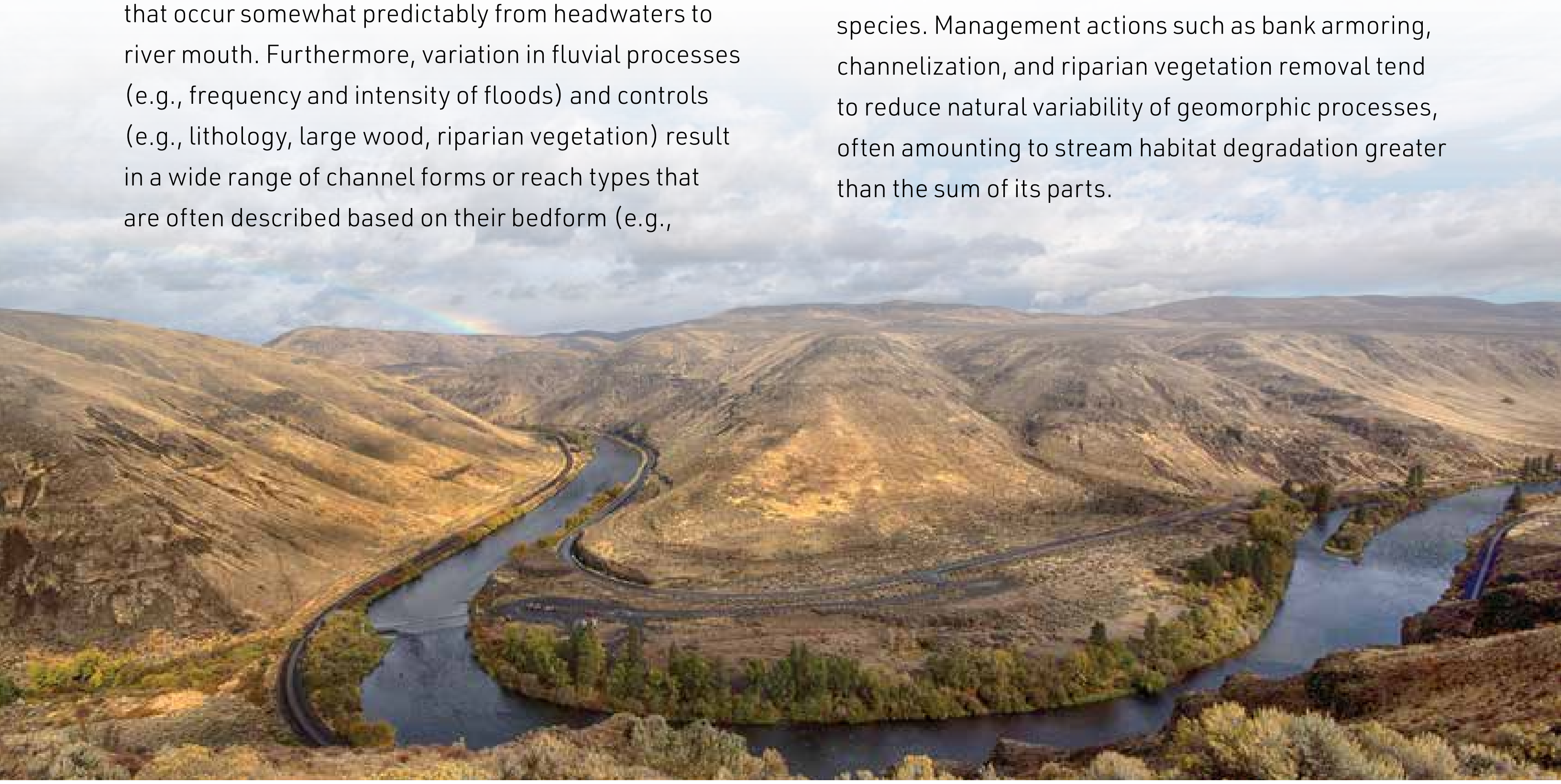
Chapter 2. Stream Morphology

By: Jane B. Atha, Mike Liquori, Kirk L. Krueger, George F. Wilhere, and Timothy Quinn

2.1. Introduction

The composition, structure, and functions of lotic systems are largely due to dynamic interactions of water, sediment, wood, and riparian vegetation. The nature of these interactions is affected by topography (e.g., channel slope), flow regime, sources and characteristics of sediment, the size, strength, and density of riparian vegetation (Naiman and Bilby 1998), and the actions of some animal species, most notably the American Beaver *Castor canadensis* and salmon. The influences of water and channel slope on sediment results in channel patterns (i.e., straight, meandering, braided) (Leopold and Wolman 1957) and geomorphic zones (i.e., source, transport, depositional) (Schumm 1977) that occur somewhat predictably from headwaters to river mouth. Furthermore, variation in fluvial processes (e.g., frequency and intensity of floods) and controls (e.g., lithology, large wood, riparian vegetation) result in a wide range of channel forms or reach types that are often described based on their bedform (e.g., slope, sinuosity, distribution of sediment sizes, etc.) (Montgomery and Buffington 1997). These channel forms can be useful descriptors of aquatic species' habitats. Tributary junctions (Kiffney et al. 2006), natural disturbances (Frissell et al. 1986), and artificial structures (Stanford and Ward 2001) can create discontinuities in these general patterns.

The composition and structure of riparian vegetation affects channel morphology, which in turn affects riparian vegetation (Hupp and Osterkamp 1996; Corenblit et al. 2007; Osterkamp and Hupp 2010; Gurnell 2014). Additionally, channel morphology and the processes that shape it can be impacted by human activities (e.g., levees, dams, Beaver trapping), usually resulting in loss of habitats, reduced habitat diversity, and diminished habitat functions for aquatic species. Management actions such as bank armoring, channelization, and riparian vegetation removal tend to reduce natural variability of geomorphic processes, often amounting to stream habitat degradation greater than the sum of its parts.



Yakima River/Justin Haug, WDFW

Knowledge of the fluvial processes that affect riparian and aquatic systems is important for making sound policy and management decisions. This chapter provides an overview of water and sediment in fluvial processes, their interactions with riparian vegetation, and potential alterations of fluvial processes by human activities. We first describe a general conceptual model for considering how physical processes affect channel morphology. Next, we describe stream flow and sediment processes in more detail. We then consider the watershed-scale context that directly influences fluvial processes. Lastly, we review impacts of human activities on stream functioning, introduce the "riverscape" concept for the management of fish habitats, and describe some physical stream habitat associations of herpetofauna. Some related topics are discussed in other chapters. For example, see Chapter 3 for a description of fluvial process interactions with large wood and Chapter 8 for a thorough examination of influences and management that occur at the watershed scale.

2.2. Conceptual Context

The concept of equilibrium is one of the most important ideas in fluvial geomorphology (Thorn and Welford 1994; Grant et al. 2013). The equilibrium concept posits that rivers tend to work toward long-term equilibrium (Gilbert 1877 as cited by Grant et al. 2013). *Dynamic equilibrium* describes a state of relative stability that results from the interaction of opposing forces (e.g., flowing water and substrate resistance). That is, over the long term and assuming that process drivers like climate and land use remain unchanged (see Chapter 8), overall progression of the composition and structure of stream reaches in a watershed remains largely unchanged. A stream network may undergo significant changes during an extreme flood event, and yet return to equilibrium as smaller flows rework the sediment distributed by floodwaters to re-establish roughly the same relative abundance of stream reach types that existed prior to the flood event.

Lane (1955) conceptualized equilibrium within a stable channel as a balance among stream discharge, channel slope, sediment load, and sediment size (Figure 2.1). Lane's conceptual model was described by Heede (1980) as:

$$Q_s D \propto Q_w S$$

where Q_S is sediment load (i.e., sediment discharge), Q_W is water discharge, D is sediment size, and S is channel slope.

What remains "in balance" over time are the processes of streambed degradation (erosion) and aggradation (sediment deposition). If an increase occurs in stream discharge, for instance, then the resulting increase in stream power can cause an increase in streambed degradation. The channel "adjusts" by decreasing channel slope, which reduces stream power and causes a return to the balance between degradation and aggradation. If a shift in stream discharge or sediment supply occurs, then the channel returns to equilibrium through adjustments in one or more other variables. However, persistent changes in one or more variables of sufficient magnitude may exceed a threshold that causes an abrupt shift to a new equilibrium state (Schumm 1979).

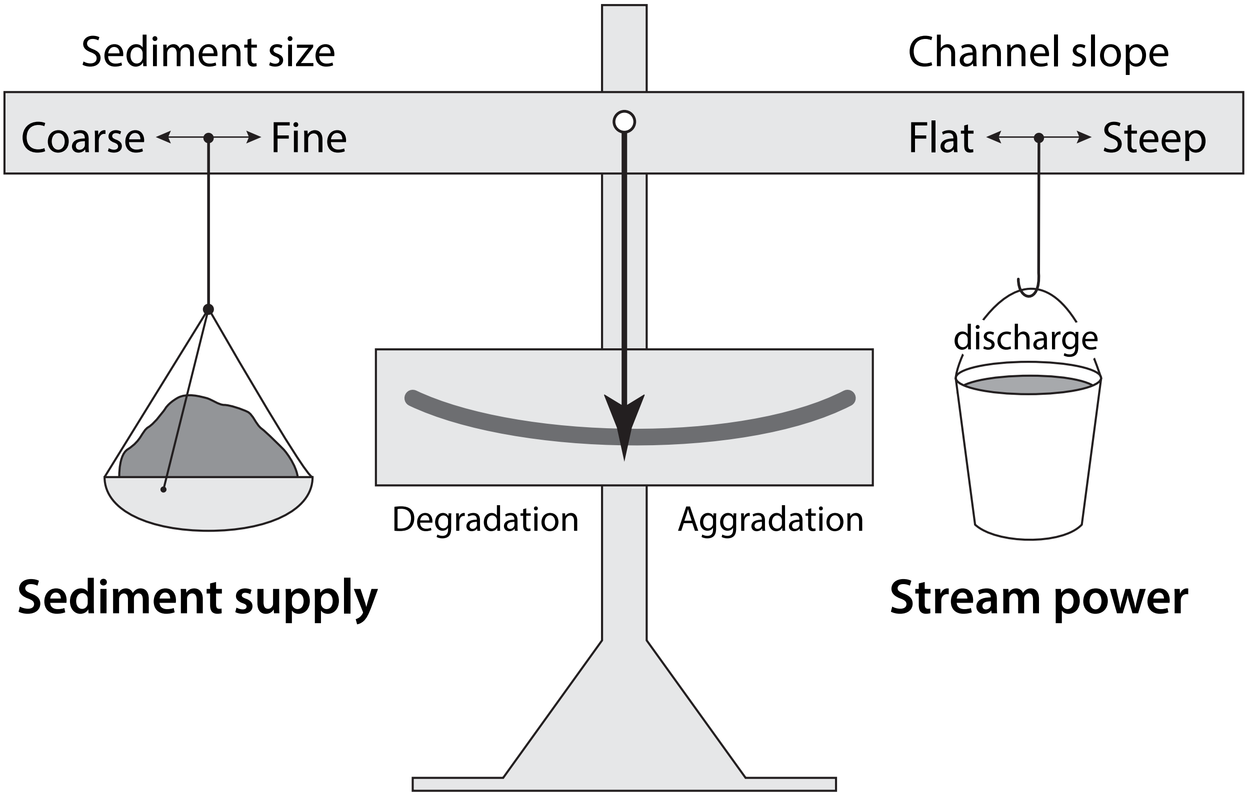


Figure 2.1. Lane's Balance illustrating the concept of channel adjustment (i.e., vertical aggradation or degradation of the streambed) in response to changes in water or sediment yield (from Dust and Wohl 2012).

While Lane's Balance is a useful conceptual model, it does not address complexity at the watershed scale nor the interactions between fluvial processes and riparian vegetation that affect channel morphology (Corenblit et al. 2007; Gurnell 2014) (Figure 2.2). These interactions are reciprocal and feedbacks exist between fluvial processes and vegetation composition and structure (Hupp and Osterkamp 1996; Osterkamp and Hupp 2010). Vegetation influences channel form and channel forming processes help shape riparian plant communities. Human activities that alter these interactions can cause significant changes to channel morphology and fish habitats.

Many geomorphic and ecosystem processes are coupled and occur in parallel at similar spatial and temporal scales. This results in mutual dependence between processes (Renschler et al. 2007). While such processes have received much attention from the scientific community in the last fifteen years, a theoretical synthesis within a holistic conceptual framework is still needed to address how riparian management interacts with this complex system (Smith et al. 2002; Stallins 2006). Interdisciplinary research of these interactions could result in better predictability of impacts caused by management. From an ecosystem management perspective, the choice of spatiotemporal boundaries—often dictated by geomorphic processes—is of profound importance to conceptualization of the system and its ecologic processes (Post et al. 2007).

We frame the hydrologic and physical processes as they shape the riverscape from the context of equilibrium and mutual dependence among water, sediment, and vegetation.

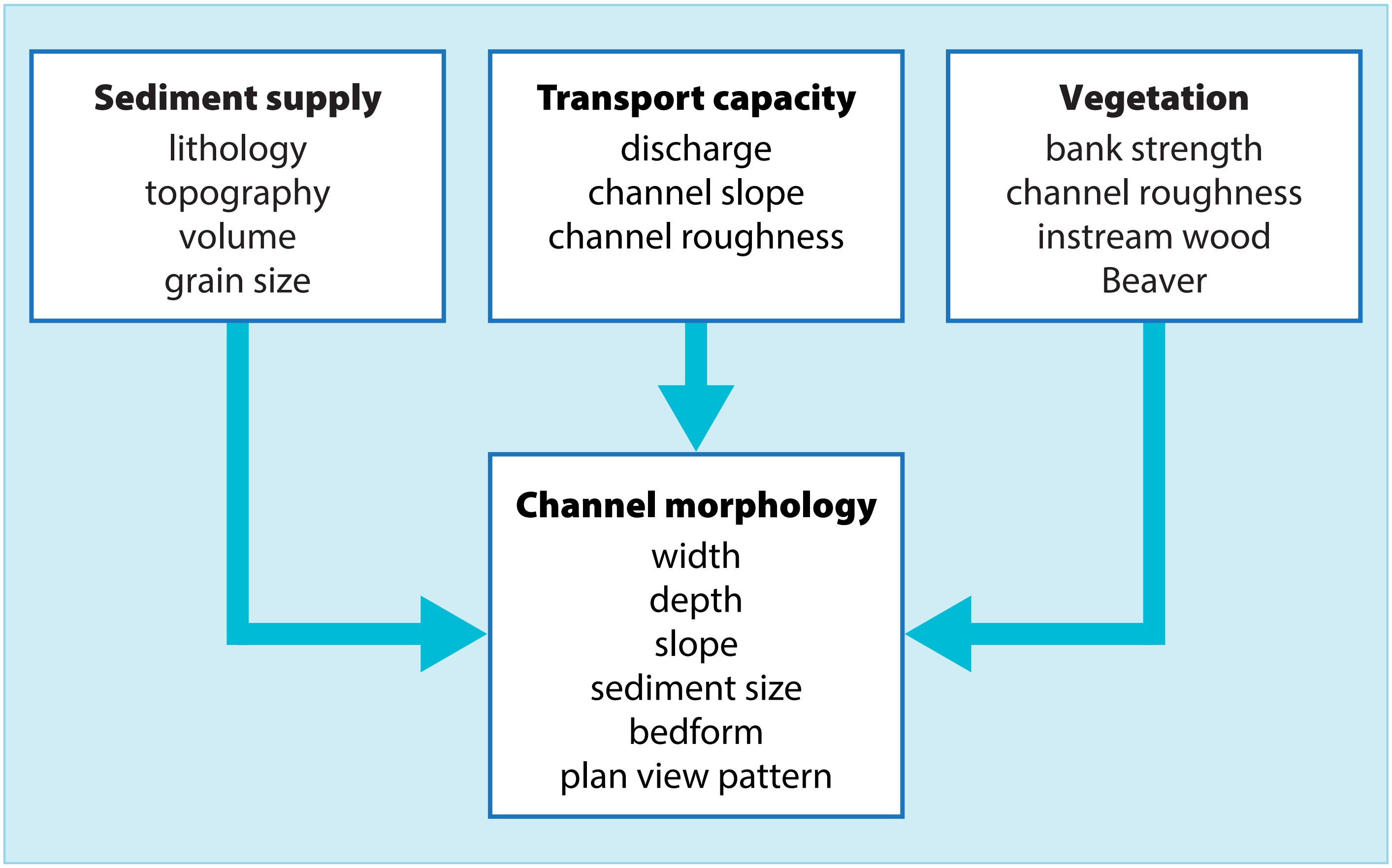


Figure 2.2. Influences on channel morphology (modified from Naiman and Bilby 1998).

2.3. Streamflow Processes

The pathways that water takes to reach a river exert a strong influence on the amount and timing of streamflow in the channel (Figure 2.3). Differences in the relative contributions of these pathways among locations and seasonal variability (e.g., presence or absence of snow) in environmental conditions produce a wide range of stream flow responses to precipitation events.

The fate of precipitation that is not lost through evaporation and transpiration is highly dependent on soil cover. Rain or melted snow that infiltrates the soil can flow downslope in the vadose zone (unsaturated), above the water table as through flow, or below the water table as groundwater in the phreatic zone (saturated).

When precipitation rates exceed soil infiltration rates (Horton 1933) or when local soils are saturated (Dunne and Black 1970a, 1970b; Dunne 1978), water moves downslope as surface runoff. Surface runoff or overland flow remains outside the confines of a channel, and is most common where vegetation is sparse. Human activities that compact soil or destroy permeable, near-surface soil layers reduce infiltration capacity and promote overland flow. A second type of runoff, saturated overland flow, occurs at convergent zones (such as hillslope concavities) and is a combination of direct precipitation onto saturated areas and return flow from the subsurface as it becomes saturated (Wohl 2014) (Figure 2.3).

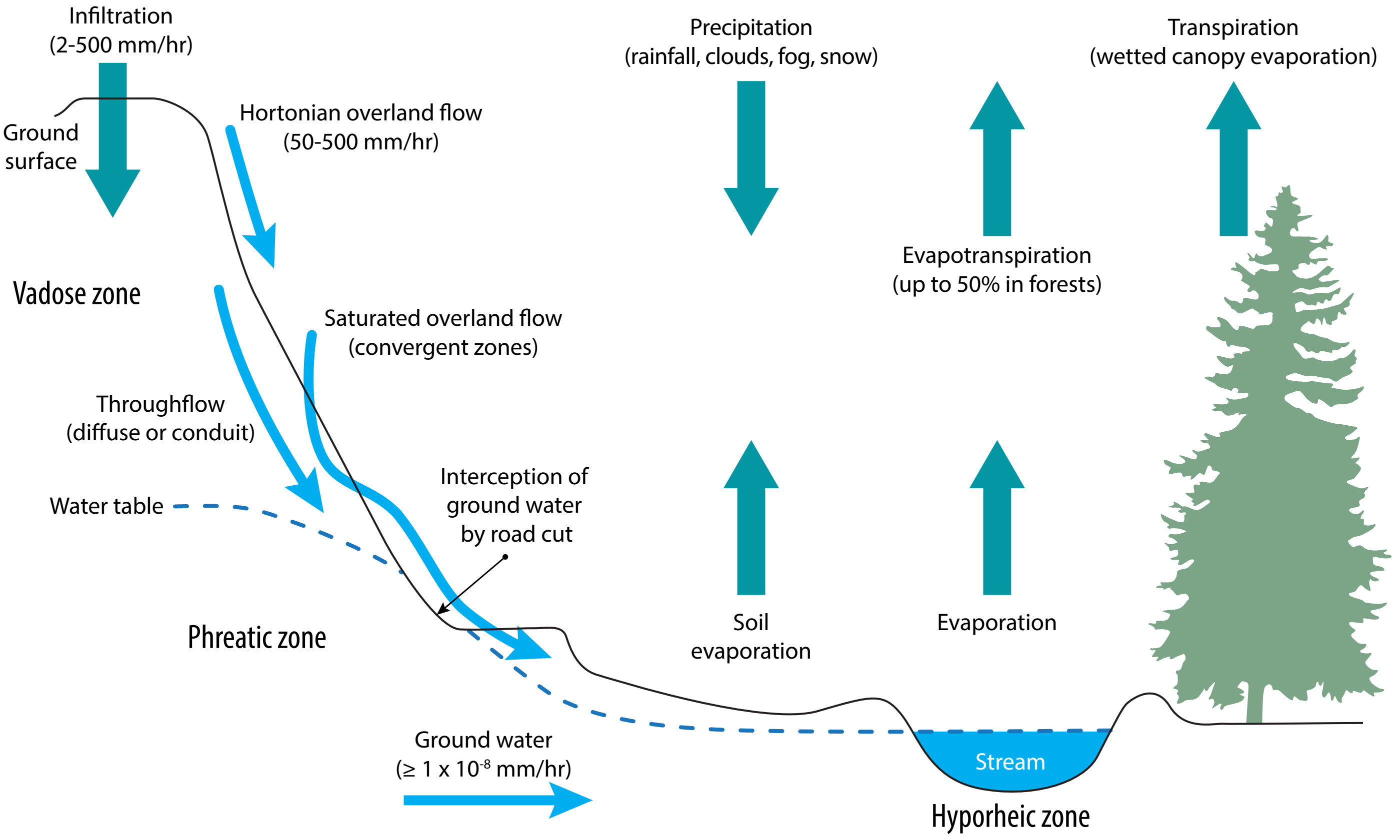


Figure 2.3. Illustration of different types and rates of downslope movement of water on a hillslope (modified from Ziemer and Lisle 1998, and Wohl 2014).

Runoff originates from variable source areas in response to precipitation and soil saturation conditions (Betson 1964; Hewlett and Nutter 1970). Conceptually, the amount of runoff within a single small drainage basin can be highly varied in space and time because many factors influence water infiltration and movement. Much of the variability in areas contributing water to the channel reflects the groundwater levels and activation[1] thresholds for lateral subsurface flow existing immediately after a precipitation event (McDonnell 2003).

Riparian vegetation influences preferential flow pathways in both surface and subsurface environments (Swanson et al. 1998; McDonnell 2003) and the manner in which surface flows create and maintain channel and floodplain morphology. During periods of active runoff, riparian areas quickly become saturated, and become the first areas within the watershed to contribute runoff into the channel (McDonnell 2003). Riparian areas account for the majority of the runoff at the beginning of a flood event, whereas hillslopes contribute more as the runoff tapers off at the end of an event. Flow resistance provided by riparian vegetation and related organic material slows water velocities, reduces peak discharge, and lengthens a flood's duration (Tabacchi et al. 2000; Nilsson and Svedmark 2002).

In a reciprocal manner, stream hydrology plays a large role in shaping riparian vegetation communities. The rate at which the river water surface lowers after a spring freshet influences seedling germination and recruitment (Mahoney and Rood 1998). Thus vegetation species composition and age structure vary due to flow frequencies that determine sediment mobilization and effective discharge (Pike and Scatena 2010).

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[1] Activation of subsurface water means that subsurface water is flowing, i.e., the velocity deviates significantly from zero. Activation threshold often refers to the amount of precipitation needed to activate subsurface flow.

Interactions between stream water and groundwater occur in the hyporheic zone. In other words, the *hyporheic zone* is the ecotone between a surface stream and deeper groundwater (Boultan et al. 1998). The extent of the hyporheic zone is determined by the volume of sediment stored within the channel and active floodplain (Moore and Wondzell 2005). Two-way exchanges of water occurring in the hyporheic zone are important in regulating hydrological, biogeochemical, and thermal processes (Krause et al. 2011), which strongly influence the volume, chemistry, and temperature of streamflow. Along this ecotone, upwelling subsurface water supplies the stream ecosystem with mineral nutrients while downwelling stream water provides dissolved oxygen and organic matter to organisms inhabiting the interstitial spaces of alluvial sediments (Boultan et al. 1998). The hyporheic zone can account for a fifth of the invertebrate production in an aquatic ecosystem (Smock et al. 1992). Water flows laterally from the hyporheic zone into alluvial sediment deposits, and can extend out more than 1.2 miles (2 km) in wide valleys and to depths of 33 ft (10 m) (Stanford and Ward 1988). Hyporheic flows can also affect riparian vegetation, although the interactions between riparian communities and hyporheic conditions are not well understood (NRC 2002).

Cape Creek, Lane County, Oregon/Jane Atha, WDFW

2.4. Erosion and Sedimentation Processes

2.4.1. Sediment Sources and Delivery

Sediment sources and delivery within a watershed can vary significantly depending upon geology, climate, landform, soil erodibility, vegetation, and the magnitude and frequency of dominant disturbances. Delivery of sediment to channels by surface erosion is generally low in undisturbed watersheds, but can vary greatly by year (Swanston 1991). Annual differences are caused by variation in weather patterns, changes in exposed surface area, and changes in the amount of erodible material. Sediment can move downslope in aggregates through mass wasting (mass movement, mass failure, or slope movement) categorized as landslides, debris flows, or heaves. Mass wasting events are typically seasonal as a function of soil moisture and freeze-thaw processes (Hales and Roering 2009). Soil on a hillslope will remain stable if the sum of the applied shear stresses does not exceed the sum of the shear strengths of slope materials (soils, roots, etc.). Mass wasting is initiated when shear stress on a material exceeds its shear strength threshold (Ritter et al. 2002). The volume of material generated from hillslope mass wasting events can be very large, and is often the predominant source of sediment and large wood delivered to streams in the Pacific Northwest (Swanson et al. 1982).

Sediment can also move downslope through gradual, diffuse processes such as rainsplash and rills that form from overland flow. Rainsplash takes place when raindrops loosen the soil particles making them more

Skagit Valley agriculture/Jane Atha, WDFW

susceptible to entrainment by overland flow (Furbish et al. 2009; Dunne et al. 2010). Overland flow can be concentrated into small, narrow, shallowly incised micro-channels that are carved into hillslope soils (Selby 1993). These channels, termed rills and gullies (with gullies being deeper) can form effective conduits for sediment erosion into stream channels (Wohl 2014).

Sediment delivered from riparian areas can be generated by mechanisms other than direct streambank erosion. Riparian sediment sources can be associated with treefall that results in the uprooting of trees (root throw; Hairston-Strang and Adams 2000; Liquori 2006). The primary causes of treefall include wind along newly created edges in riparian forest (Liquori 2006; Reid and Hilton 1998), undercutting streambank erosion, and large-scale disturbances such as fire or pest infestation. Root throw can provide direct sediment inputs to the channel (Lewis 1998; Reid and Hilton 1998; Gomi et al. 2005). Harris (1977) found that windthrow-related sediment production along a stream in coastal Oregon was responsible for 33 tons of sediment per mile of stream per year (18.6 metric tons per kilometer per year) (as cited in Gomi et al. 2005).

Riparian vegetation can effectively trap and store sediment transported downslope from overland flow (Naiman et al. 2010). This topic is fully explored in Chapter 5, Pollutant Removal. Once the sediment is deposited in riparian areas, plant roots facilitate stabilization and continued storage of sediment (Allmendinger et al. 2005; Tal and Paola 2007).

2.4.2. Instream Sediment Transport and Deposition

Sediment that has entered the stream channel can be transported downstream as dissolved load, suspended load, or bed load. Dissolved load carries sediment within solution, which can be substantial in some rivers (Knighton 2014). The concentration of solute is highest in water entering the channel through subsurface pathways because slower rates of subsurface water movement allow for longer reaction times with the surrounding environment. Solute concentrations through subsurface pathways tend to lessen with greater discharge (Wohl 2014). Suspended load refers to sediment carried as suspended particulates in the water, and originates predominantly from bank erosion and surface erosion from upland areas. The finest size fractions of suspended transport, known as wash load, are more likely to be supply limited, i.e., the amount in transport is limited by the amount available. In contrast, the coarser bed load, which consists of larger rocks that bounce along the channel bed, is typically limited by transport capacity of flowing water (stream energy). Bed load may not get suspended; it can roll and slide during low flows, saltate (bounce) at intermediate flows, and move as sheetflow during high flows (Wohl 2014). Bed load transport processes are surprisingly complicated because many factors, some of which interact, determine the threshold for movement. These factors include the range of sediment sizes, their spatial arrangement, and channel form within a reach (Knighton 1984).

Bedforms are bed undulations that result from sediment transport and deposition. Montgomery and Buffington (1997) classified stream channels based on common bedforms (i.e., cascade, step-pool, plane-bed, pool-riffle, and dune-riffle), their dominant sediment transport process, and behaviors of debris flows and instream wood. The role that sediment plays in aquatic habitat can differ dramatically among those classes. For example, sediment in headwater channels (often classified as cascade or step-pool) tends to be stored behind boulder and wood obstructions. In lower gradient channels, sediments become sorted into pools and riffles depending on instream wood load, gradient, and a wide variety of other factors (Benda et al. 2005; Gomi et al. 2005).

All bedforms are depositional features. Deposition occurs when the flow or shear velocity in the stream falls below the settling threshold velocity of the sediment particle. Settling thresholds are lower than entrainment thresholds, and deposition can be localized or extended across a channel length depending on fluctuations in flow velocity (Wohl 2014). Riparian vegetation can be especially instrumental in facilitating deposition within channels and along banks (Tal et al. 2004). Establishment of riparian vegetation can initiate a positive feedback process as plants trap and stabilize sediments and organic matter that provides sites for germination of plants, which can then reinforce the development of floodplains, instream islands, and other landforms.

Erosional forces in the streambed can be increased due to topographic constrictions or locally steep gradients. Where hydraulic forces overcome surface resistance, channel incision can occur (Wells et al. 2009). Channel incision that occurs at the channel head, known as a headcut, results in vertical faces that separate upslope unchanneled environments from downslope channels. Headcuts that destabilize or incise an otherwise stable channel bed result in significant increases in sediment yield (Meyer et al. 1975; Bryan 1990). Headcuts can also occur as an upstream-migrating "step" in the channel bed that forms as a result of hydraulic scour, often from manipulations to the channel or flow conditions (e.g., culverts, diversions, etc.). This type of headcut can migrate upstream and change channel form for long distances.

Establishment of riparian vegetation can initiate a positive feedback process as plants trap and stabilize sediments and organic matter that provide sites for germination of plants, which can then reinforce the development of floodplains, instream islands, and other landforms.

A combination of field, experimental, and theoretical research has dramatically advanced our understanding of sediment transport dynamics as well as the role of vegetation in stream morphology. Sophisticated flume experiments and computational models have confirmed the importance and complexity of interactions among water, sediment, wood, and vegetation (Gurnell 2014). Recent modeling efforts have placed these phenomena on solid theoretical footings and have allowed researchers to explore scale dependence. We know much less about these interactions at small scales (single plant), however, understanding smaller scale phenomena is crucial to fully understanding longer-term and larger-scale geomorphic form and process (Gurnell 2014).

Bank erosion along the upper Chehalis River/Jane Atha, WDFW

2.4.3. Streambank Erosion

Streambank erosion is a geomorphic process that is integral to a watershed's natural disturbance regime and necessary for long-term ecological sustainability (Florsheim et al. 2008). Bank erosion is a key process for maintaining the structural diversity of aquatic and riparian habitats; it initiates community succession that promotes ecological diversity of riparian areas, and contributes coarse sediment to streambeds that is an essential habitat element for benthic invertebrates and spawning salmon. Bank erosion commonly occurs on the outside of river bends, but it can also occur in straight channels where changes in water or sediment supply cause channel incision. Instream large wood or mid-channel gravel bars that divert flows toward a bank (Florshiem et al. 2008) can also cause bank erosion. Local bank erosion is a naturally occurring process that is often indicative of a channel in dynamic equilibrium. Despite this fact, bank erosion at any scale is commonly misinterpreted as a sign of channel instability. Human disturbances can increase rates of bank erosion, and thus, understanding whether bank erosion is triggered by human activities or natural geomorphic processes is important for site-scale management (Polvi et al. 2014).

Two types of processes result in bank erosion: fluvial erosion and mass wasting (ASCE 1998). Fluvial erosion is the separation of sediments from a streambank's surface by the forces of flowing water. Fluvial erosion may destabilize riparian vegetation by exposing plant roots or undercutting vegetation (Florsheim et al. 2008). Fluvial erosion can eventually lead to mass wasting ("bank failure"). Mass wasting at streambanks is caused by scour of the streambed and bank toe that increases bank height and angle. When gravitational forces on the bank (weight of soil, water, and overlying vegetation) exceed the shear strength of the bank material, a portion of the bank collapses along a failure plane (Simon et al. 2000). Both types of bank erosion occur mainly during peak flows, but the association between flow magnitude and amount of bank erosion varies greatly among

Building in danger from streambank erosion on Newaukum River/Kaysie Cox, WDFW

watersheds. Wolman (1959, as cited in ASCE 1998), for instance, reported that significant bank erosion on a creek in Maryland occurred more than ten times per year during relatively small but frequent peak flow events. Studies of other alluvial systems have found that significant bank erosion was caused mostly by large floods with recurrence intervals of decades or more (ASCE 1998).

The stability of streambanks is influenced by soil characteristics, groundwater, and vegetation (Hickin 1984; NRC 2002). Mass wasting failure mechanics, for example, are quite different for cohesive and noncohesive soils (ASCE 1998). Adjacent surface waters or ground water affects pore water pressure, and positive pore water pressure destabilizes banks by reducing cohesion and friction amongst soil particles. Water in soil also adds mass, thereby increasing gravitational forces acting on a streambank. Bank failure often occurs shortly after flood waters recede because soils are at or near saturation and the laterally confining hydraulic pressure of the floodwaters decreases to zero (Rinaldi et al. 2004).

Vegetation may have mechanical and hydrologic effects on bank stability, and these effects can be stabilizing or destabilizing (Simon and Collison 2002; Langendoen et al. 2009; Pollen-Bankhead and Simon 2010). Soil is generally strong in compression, but weak in tension. Woody and herbaceous plant roots are strong in tension, but weak in compression. Consequently, the root-permeated soil of streambanks behaves as a composite material with enhanced strength (Simon and Collision 2002; Pollen and Simon 2005). However, the additional bank strength provided by roots is species dependent (Polvi et al. 2014). Simon et al. (2006), for instance, found that Lemmons's Willow *Salix lemmonii* provided an order of magnitude more root reinforcement of streambanks than Lodgepole Pine *Pinus contorta*. Dense root networks also physically restrain or bind soil particles. In addition, exposed roots on the bank surface increase channel roughness, which dampens stream flow velocities, thereby reducing fluvial erosion (Griffin et al. 2005; Gorrick and Rodríguez 2012). In fact, fluvial erosion of well-vegetated banks is 10 to 100 times less than erosion of unvegetated banks (ASCE

1998). Reduction of stream velocities by roots may also cause sediment deposition, which further stabilizes streambanks.

Vegetation has destabilizing mechanical effects as well (ASCE 1998; Simon and Collison 2002). The weight of vegetation adds a "surcharge"[2] load to soil. Hence, the weight of vegetation increases the vertical shear stress near a streambank and adds to the lateral earth pressure[3] at the streambank surface, however, these effects can be minimal relative to soil's weight (Simon et al. 2006). Additionally, tall, stiff vegetation may impose destabilizing forces on streambanks during windstorms (ASCE 1998).

The hydrologic effects of plants on streambank stability are those that influence soil moisture. Through canopy interception, vegetation reduces the amount of precipitation that reaches soil. Transpiration, which extracts water from the soil column within the root zone, creates negative pore water pressures (matric suction) that increase soil shear strength (Simon and Collison 2002). Plants destabilize streambanks by facilitating infiltration of water into soil via flow pathways ("macropores") created by live and decayed roots. In addition, stemflow tends to concentrate precipitation around the base of stems, creating higher local pore water pressure. The net hydrological effect of vegetation can be significant. Simon and Collision (2002), for instance, found in their study conducted in northern Mississippi that following a very dry period the stabilizing hydrological effect of tree cover was two times greater (220%) than tree cover's stabilizing mechanical effect. However, this result was reversed following a wet period; the stabilizing mechanical effect of tree cover was 159% of tree cover stabilizing hydrological effect.

The bank stability function of riparian vegetation has important implications for riparian area management. FEMAT (1993:V-27) created a generalized curve of root strength versus distance from channel based on expert opinion informed by scientific literature. The radius of Douglas-fir *Pseudotsuga menziesii* root networks are highly correlated with tree crown radius (Smith 1964).[4] Hence, FEMAT (1993) assumed the contribution of root strength to maintaining streambank integrity declines at distances greater than one-half a crown diameter. Using linear regression, Roering et al. (2003) derived a power-law relationship for coniferous trees between root network radius and tree diameter. This relationship suggests that large diameter conifers (36 to 48 inches [91 to 122 cm] diameter breast height) have root network radii that measure 16 to 33 ft (5 to 10 m). Hence, in an undisturbed old-growth riparian forest, the full contribution of root strength to streambank stability is provided by trees within approximately 33 ft (11 m) of the streambank.

2.4.4. Lateral Channel Migration

Lateral channel migration affects sediment erosion and deposition, local topographic relief, flood inundation patterns, river planform, alluvial architecture, and riparian vegetation patterns, which in turn results in a complex arrangement of aquatic and riparian habitats across a river's floodplain (Naiman et al. 1993). Lateral channel migration occurs through the processes of meander bend development, avulsions, and channel widening and narrowing (Rapp and Abbe 2003). These processes occur through fluvial erosion of intact bank material and mass failure of streambanks under gravity, followed by mobilization and transport of the disturbed material (Burckhardt and Todd 1998).

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[2] A surcharge load is any load imposed upon the soil surface close enough to a streambank to cause lateral pressure to act on the bank. "Surcharge" denotes an extra force in addition to the weight of the soil.

[3] Lateral earth pressure is the pressure (i.e., force per area) that soil exerts in the horizontal direction.

[4] The ratio of root spread to crown width for small Douglas-fir and Western Hemlock *(Tsuga heterophylla)* (mean dbh about 13 inches [33 cm]) was reported by Smith (1964) to be about 1. Data in Eis (1987) for these same tree species suggest a ratio of about 2.

One mechanism of channel migration is meandering, which is one of the most common river-channel patterns (Güneralp and Marston 2012). Given time, and lacking physical obstructions, a natural unimpeded meandering channel can swing and shift across its valley in the downstream direction, completely disturbing the floodplain (Schumm 1977). The floodplains of meandering rivers are low-lying depositional features that store large volumes of sediment, and reflect the history of erosion and deposition of a river occupying the valley floor (Dunne and Leopold 1978). The time needed for a channel to migrate and occupy its entire floodplain is expressed as a "floodplain turnover rate," which may take tens to thousands of years in western Washington. Beechie et al. (2006) found, for example, that braided channels in rivers of the Cascades and Olympic mountains had a turnover rate of 25 years. In contrast, O'Connor et al. (2003) estimated turnover rates for two large rivers on the Olympic Peninsula to be approximately 200 to 2000 years. The channel migration rates for the rivers in O'Connor et al.'s study ranged from 13 ft (4.0 m) to 42 ft (12.7 m) per year.

Sediment erosion and deposition patterns that result in channel migration are caused by complex flow patterns along meander bends. Water tends to develop a horizontal swirling flow as a result of greater roughness along the banks than in the center of the channel. This swirling flow, termed "helical flow", is generated by the super elevation of the water's surface as it flows along the outside of a bend (Callander 1978). Helical flow creates large cross-stream variation in velocity (Dietrich et al. 1984), and the reduced velocity and bed shear stress at the inside of a bend creates pointbar deposition. In addition, pointbars deflect flow laterally toward the cutbank, creating topographically driven secondary flow that further induces an outward velocity shift toward the cutbank (Güneralp and Marston 2012).

Movements of individual meanders are described as: 1) simple migration (or translation), which is a downstream shift without significant shape change; 2) extension (or growth), where the amplitude of the bend extends by migrating across the valley; 3) rotation, in which the bend axis changes orientation; and 4) lobing and compound growth, where the bends become less regular and symmetrical (Hooke 1984). Individual bends along a meandering channel can have different styles and rates of migration, and typically deform and become asymmetrical with migration. Meander migration that increases the amplitude and tightness of bends can lead to a cutoff channel that creates a shorter flow path across the inside of a pointbar or a neck cutoff at the base of the bend. The increase in channel gradient from the cutoff causes it to become the main channel, and the longer flow path becomes a secondary or overflow channel on the floodplain. Secondary channels may function as side channels that are important rearing habitat for juvenile salmonids (Rosenfeld et al. 2008) or

Chehalis River meander bend/Jane Atha, WDFW

Mid-channel bar/Jane Atha, WDFW

oxbow lakes, which are habitat for amphibians (Henning and Schirato 2006). The increased habitat diversity for aquatic and riparian organisms can persist for decades to centuries depending on rates of sediment filling.

Another mechanism for channel migration is avulsion, which is the rapid abandonment of a river channel and the formation of a new river channel. Avulsions can occur with little erosion of the land between the old and new channel locations (Allen 1965; Butler 2004). An avulsion can cause rapid, dramatic shifts in channel location from one side of the valley bottom to the other, and can occur during a single flood event. Avulsions are often associated with aggrading channels, but this is not a necessary requirement (Tooth et al. 2007). Meander cutoffs, for example, are a type of avulsion not derived from aggradation. In the Pacific Northwest, wood jams often cause and mediate avulsions, acting to maintain a multiple-channel pattern. Wood jams can cause avulsions by plugging channels and diverting flow into a relict channel that then becomes the main channel (Collins and Montgomery 2002).

Along alluvial rivers, spatial and temporal patterns of riparian vegetation composition and structure are controlled by rates of channel migration and vegetation response to disturbance (Figure 2.4). As channels migrate laterally, pointbars create conditions for recruitment of early successional plant species. Over long periods, pointbar growth is followed by the succession of terrestrial plant communities. Consequently, a pointbar may support a chronological series of successional stages arranged from youngest closest to the active channel to oldest farthest from the channel (Shankman 1993). A limited number of species consistently dominate the accreting edge of pointbar landforms, while progressively more species establish as the pointbar aggrades to a higher surface elevation (Meitzen 2009). Opposite these pointbars, eroding cutbanks expose and remove previously established mature vegetation, but also open edge habitat for colonization by new plant species. Kupfer and Malanson (1993) found that cutbank edge forests have higher stem densities and greater species richness than floodplain interior forests.

Lateral channel migration and related streambank erosion processes can pose risks to human communities located near rivers and large streams. Likewise, human actions intended to limit channel migration can degrade aquatic and riparian habitats. For these reasons, geomorphologists have developed protocols for determining the channel migration zone (CMZ). A CMZ includes the outer extent of historical channels, plus the area where future channel migration could occur over some time period (e.g., 100 years). CMZ delineation considers the historical migration zone, i.e., the area encompassing all historical channels; the avulsion hazard zone, i.e., areas not in the historical record that are at risk of avulsion; and the erosion hazard area, i.e., the area at risk of bank erosion from stream flow or mass wasting over the time horizon of the CMZ (Rapp and Abbe 2003). The CMZ typically encompass floodplains and some portion of terraces (landform remnants of the former floodplain). The CMZ also includes channels and terrace banks that are at risk of mass wasting due to erosion of the toe.

Lateral channel migration produces a spatially dynamic riparian ecosystem. Because the riparian ecosystem (i.e., the riparian zone and zone of influence) is defined by proximity to water, as the active channel moves back

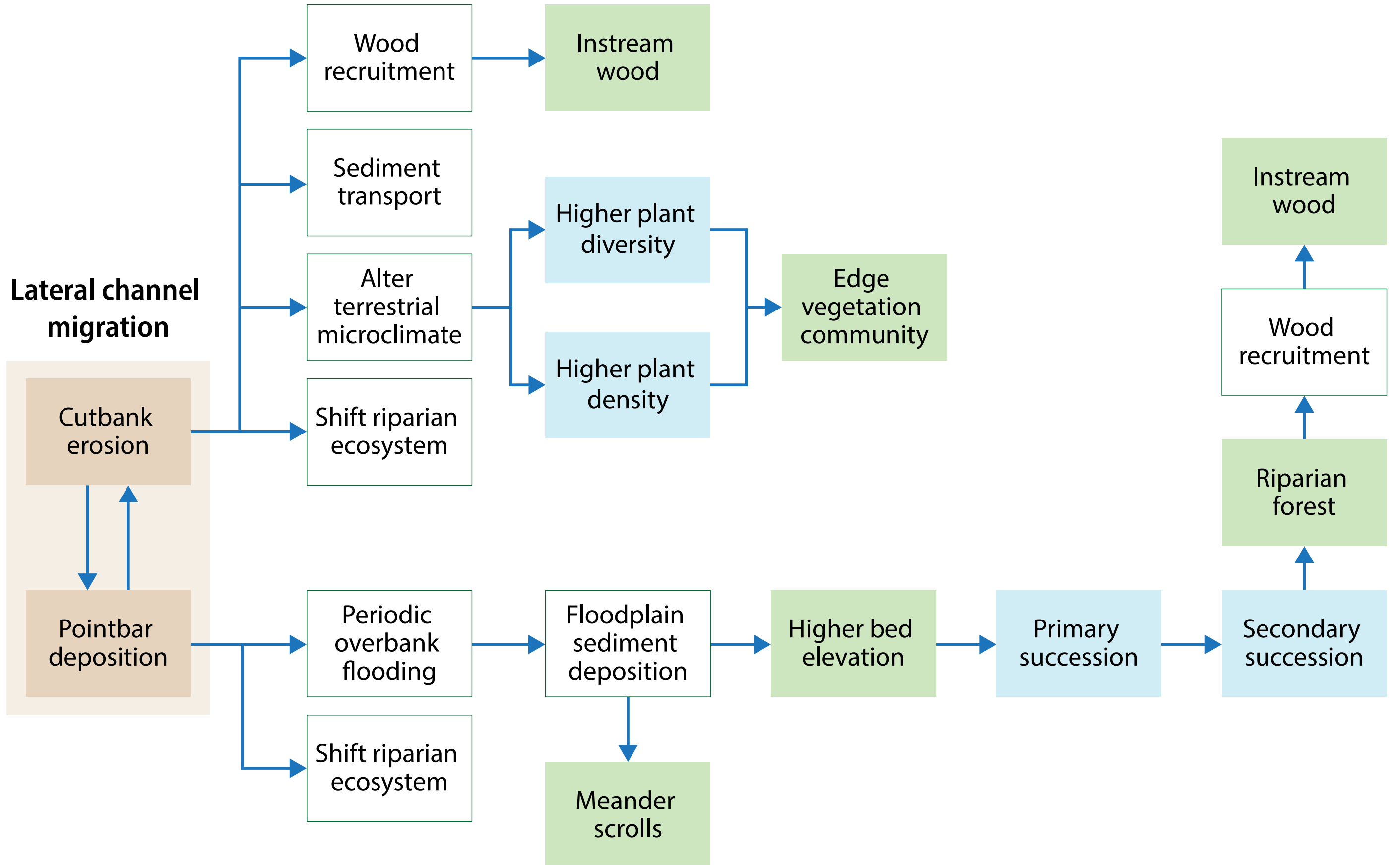
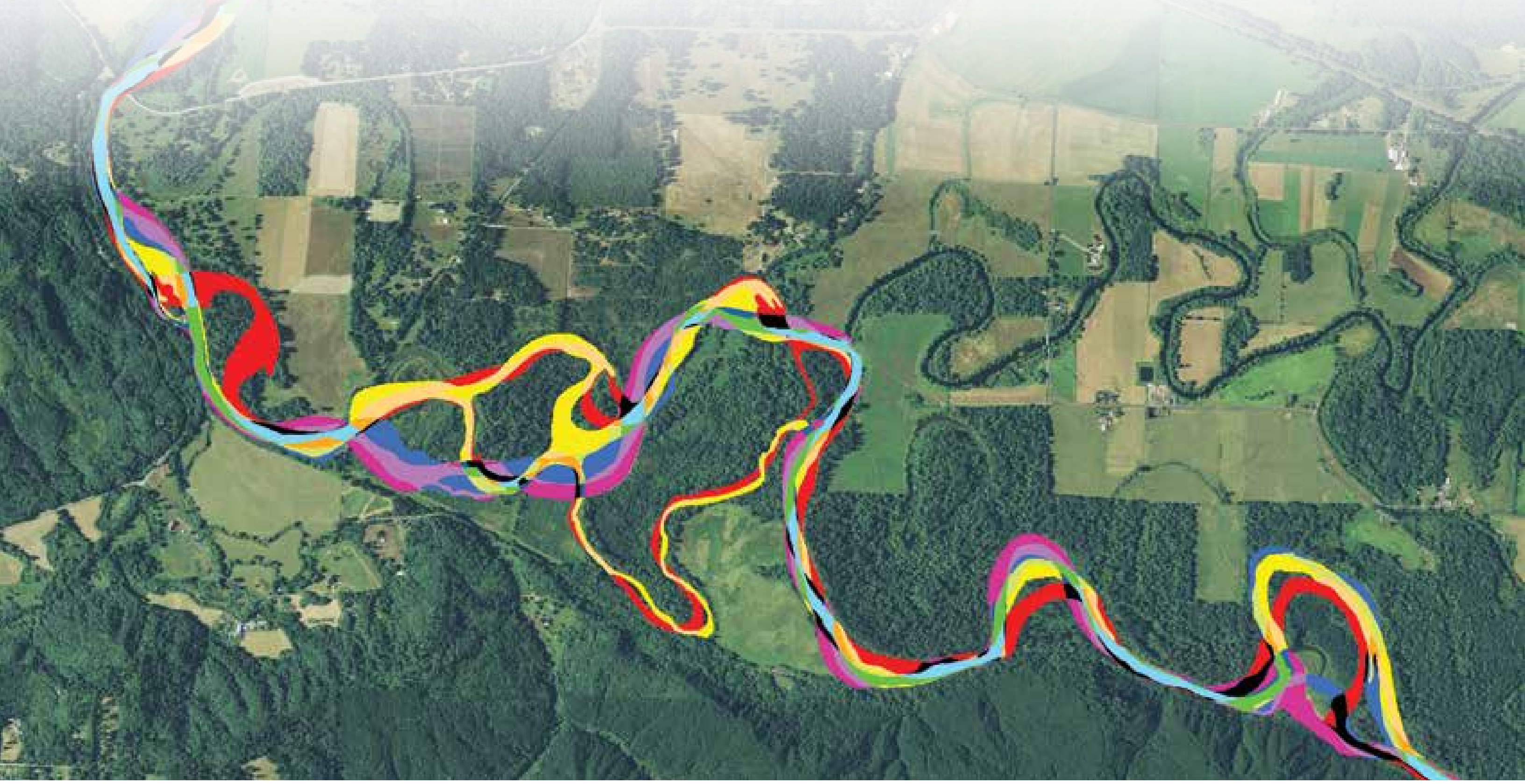


Figure 2.4. A conceptual model of lateral channel migration processes and riparian forest responses (modified from Meitzen 2009). Arrows indicate the hypothesized directions of causal relationships. Brown boxes represent proximal processes leading to channel migration. White boxes are secondary processes. Green boxes depict structures. Blue boxes are biological responses.



Chehalis River meander paths between 1938 and 2013/Ken Pierce, WDFW

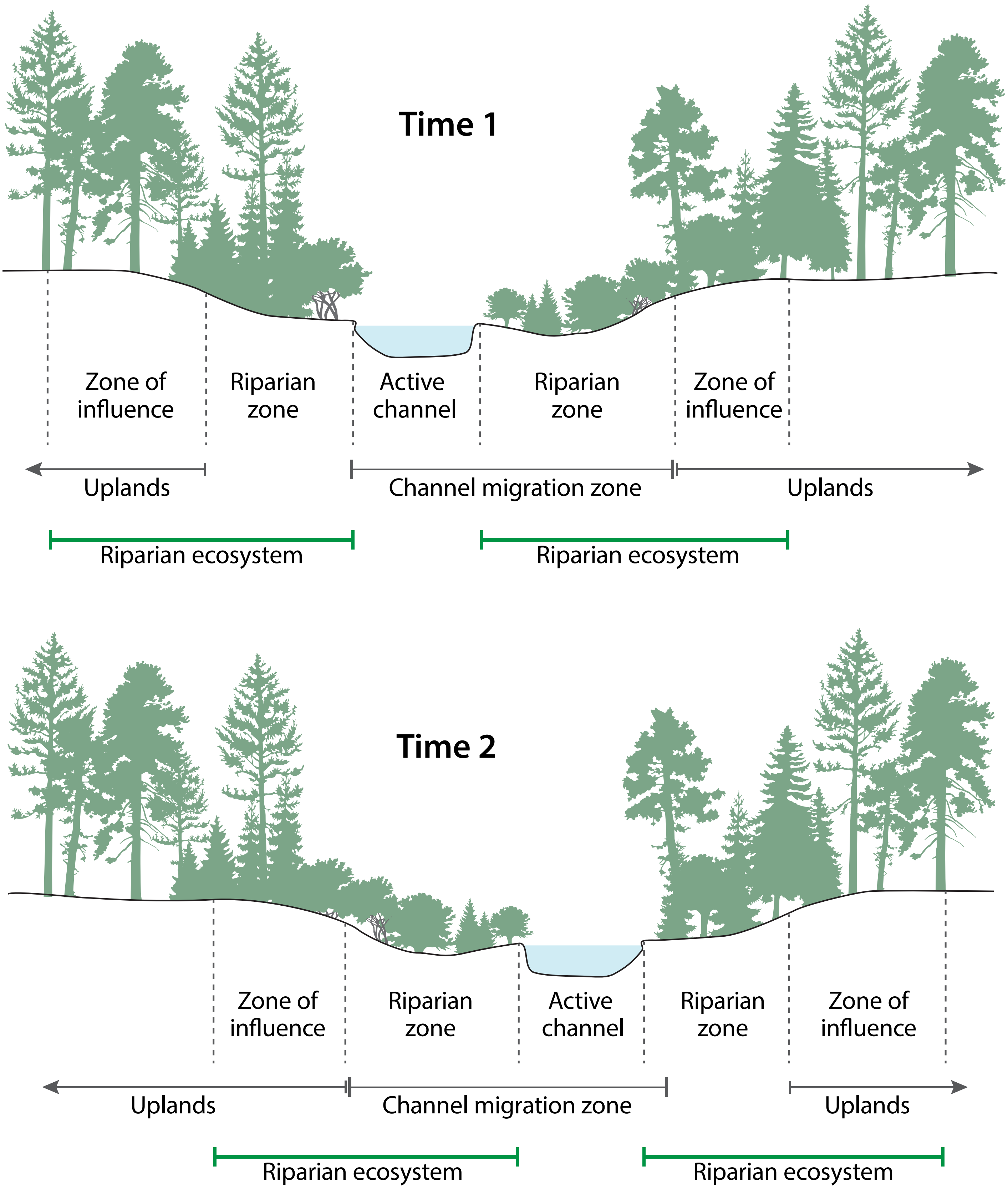


Figure 2.5. Spatial relationships between the riparian ecosystem and channel migration zone (CMZ) over time. As the active channel moves laterally within the CMZ the riparian ecosystem moves with it. Time 1 and time 2 could be separated by days or centuries.

Quinault River/Jane Atha, WDFW

and forth across the CMZ, the riparian ecosystem moves with it (Figure 2.5). Consequently, over long periods of time, a riparian ecosystem will occupy different parts of the CMZ. Furthermore, when an active channel reaches the outside limit of its CMZ, the riparian ecosystem lies outside and adjacent to the CMZ. Hence, to maintain riparian ecosystem functions, management must anticipate and protect future locations of the riparian ecosystem.

2.5. Disturbances

2.5.1. Geomorphic Process Domains

The spatial variability of sediment sources and the varying capacity of water to transport sediment result in a mosaic of different channel forms within a watershed. Channel forms can be described by attributes such as bed and bank sediment size distributions, pool and riffle depths, bankfull widths, gradient, and sinuosity. The classification of channel forms has been refined over the last 30-40 years beginning with Schumm (1977), who distinguished a headwater production zone, mid-basin transfer zone, and downstream depositional zone. This longitudinal arrangement is also reflected in more recent work that defines geomorphic *process domains.* Process domains characterize differences in the types and frequency of disturbance events, and provide a way to represent spatial differences in disturbance regimes (Montgomery 1999).

Six common geomorphic process domains have been identified for forested watersheds of Washington State: hillslope, unchanneled hollows, debris-flow channels, bedrock-fluvial channels, coarse-bed alluvial channels, and fine-bed alluvial channels (Montgomery et al. 1996; Sklar and Dietrich 1998; Montgomery 1999). These process domains are important to distinguish since they each function differently in terms of sediment dynamics and transport capacity. Note that although geomorphic process domains often occur in a regular longitudinal sequence, there are many exceptions in Washington State where this is not the case.

Hillslopes are unchannelized and are typically dominated by diffusive transport or rain-facilitated transport of sediment downslope. Unchanneled hollows are hillslope concavities that serve as sediment storage sites and source zones for hillslope mass movement (Dietrich and Dunne 1978; Montgomery et al. 2009). Debris-flow channels are dominated by mass wasting sediment dynamics but channel geometry may still be influenced by fluvial processes. Bedrock-fluvial channels have underlying exposed bedrock that limits boundary erosion. In these channels, overlying alluvium is transported by scour during higher flows, and transport capacity exceeds sediment supply, resulting in exposed bedrock. Coarse-bed alluvial channels are dominated by unconsolidated gravels, cobbles, and boulders and are more likely to be supply limited, rather than stream energy limited. Lastly, fine-bed alluvial channels have non-cohesive sand-sized sediment and are more likely to be transport limited by flow energy rather than sediment supply (Wohl 2014).

Using process domains, it is possible to identify within a watershed where different geomorphic processes, disturbance regimes, and responses to disturbance dominate, and predict riparian vegetation dynamics (Naiman et al. 2000; Polvi et al. 2011). Channel morphology correlates well with bed gradient and sediment supply (Montgomery and Buffington 1997). The differences in the balance of hydraulic forces and available sediment among these channels likely lead to differences in overbank processes that can affect vegetation establishment, growth, and mortality. For example, in higher gradient channels, energy is expended on transporting sediment downstream and is not available to reshape the riparian area.

Disturbances are relatively discrete disruptive events that change the physical environment or resource availability, and are essential for developing habitat to support native biota.

2.5.2. Natural Disturbance Regimes

From an ecological perspective, disturbances are relatively discrete disruptive events that change the physical environment or resource availability (Pickett and White 1985); thereby creating opportunities for other species to become established (Nakamura and Swanson 2003). Natural disturbances result in dynamic habitats of varying physical conditions that allow for a variety of species to coexist. Community succession theory recognizes that large-scale disturbances, such as low frequency, high magnitude floods, continually reinitiate community succession in riparian areas (Pickett et al. 1987; Corenblit et al. 2007). The cycle of disturbance and succession occurring throughout a watershed maintains compositional and structural diversity in aquatic and riparian ecosystems. Disturbances that influence streams and riparian areas include wildfire, floods, landslides, tree infestations by insects, and human-induced disturbances such as residential development, timber harvest, and road construction.

Disturbance processes are essential for developing the range of habitat conditions necessary to support native biota (Swanson et al. 1998; Tabacchi et al. 2000). For example, the coarse sediments and large wood delivered to streams via mass wasting are thought to be essential components of freshwater salmonid habitats (Reeves et al. 1995). A natural disturbance regime can be characterized by frequency (how often), magnitude (how big), duration (how long), and predictability. Wildfires of a given magnitude, for example, occur with a particular statistical probability in the absence of human manipulation of a watershed (Agee 1993). In general, smaller, frequent, and varied disturbances increase the heterogeneity of channel forms, leading to an environment that is more diverse and species rich (Kauffman and Martin 1989; Malanson 1993; Tabacchi et al. 2000; Everett et al. 2003). For example, the dynamics of stream channels that result from frequent, low intensity wildfires determine short-term patterns such as seed germination and animal foraging behavior (Hughes 1997). The influence of moderate and frequent disturbances such as fire (Wright and Bailey 1992) or insect-induced tree mortality (Mattson and Addy 1975) may lead to minor reductions in the riparian canopy, but more diverse habitat conditions that are generally beneficial for salmonids (Naiman and Bilby 1998). By contrast, disturbances that are large and infrequent tend to lead to widespread alterations that have larger and longer-lasting physical impacts. Large disturbances could cause a geomorphic threshold to be exceeded, and the resulting fluvial response could manifest a new geomorphic state with different physical conditions that support a different biological community (Hughes 1997).

Human activities can either dampen or amplify disturbance processes. While natural disturbance processes are ongoing, they are modified by human alterations (e.g., dams eliminate or dampen flood flows and land management can reduce the frequency and severity of fires). Too little or too much disturbance may have detrimental impacts on the physical structure of streams and riparian areas. For example, too little flooding can lead to cementation of the streambed or loss of riparian vegetation, especially in arid regions. Conversely, too much flooding caused by land cover change, for example, can lead to less habitat complexity due to excessive scouring of the channel. The past and current management of a site—and of its greater watershed—may alter the natural disturbance regime resulting in ecosystem states that did not exist historically.

Process domains within a watershed exhibit different disturbance regimes which manifest different aquatic and riparian ecosystems. Disturbances, and where they occur within a watershed, play key roles in structuring ecosystems. Important advances have shifted focus from single channel reaches and disturbance events to processes occurring at larger geographic extents over longer periods of time, that establish linkages between watershed-scale conditions and natural disturbance regimes (Naiman and Bilby 1998; Nakamura and Swanson 2003).

2.5.3. Human Alteration Impacts

Thus far, this chapter has largely focused on the hydrologic and geomorphic processes and their interactions with riparian area vegetation as they naturally occur within a watershed. However, human influences on channel form are ubiquitous, and therefore should be considered in management of riparian areas.

How a channel responds to human alterations of the landscape or riparian areas depends on its sensitivity and resilience (Brunsden and Thornes 1979).

Narrow riparian buffers along Badger Creek/Jennifer Nelson, WDFW

A sensitive channel reach changes in response to human alterations. A resilient channel reach also changes in response to human alterations but will return to its prior condition at some later time. The idea that some parts of a river are more or less sensitive or resilient is critical to predicting channel adjustments. For example, stream reaches with high transport capacity may adjust relatively quickly to an increase in upstream sediment yield but a similar change in sediment yield may cause significant changes in channel morphology in reaches with low transport capacity. The shape, hydraulics, and natural disturbance regime of a reach may indicate its sensitivity to anthropogenic disturbance. For example, a channel that is degrading (lowering bed elevation) may be more sensitive to riparian large wood additions than a channel that is in equilibrium. In addition, sensitivity to changes in hydrology is inversely proportional to channel gradient (Wohl et al. 2007). Hence, managers should anticipate that high gradient streams will be more sensitive to changes in stream flow than lower gradient streams. Other drivers such as the flow regime or controls such as lithology might also be used to determine a reach's sensitivity and resilience, and thereby inform management decisions.

Forestry

An extensive body of scientific literature documents the effects of forestry on stream channels (see Foley et al. 2005; Scanlon et al. 2007; and Wohl 2013 for thorough reviews). Cutting of trees and road building associated with forestry can greatly increase sediment yields and affects water yields over multiple decades as vegetation recovers (Wohl 2014). Impacts from road networks—for logging and otherwise—are numerous and are discussed below. Changes in sediment supply and water yield due to logging can alter the streamflow, stream chemistry, and channel morphology of a stream reach (Liquori et al. 2008). In addition, the resilience of channels in response to such alterations can be quite variable. For example, Madej and Ozaki (1996) found that channel recovery (defined as returning to a former bed-elevation) after a sediment wave following extensive logging in Redwood Creek basin, Northern California varied from 8 years in one creek to 15 years in another. A third channel had not yet recovered.

Clearcut hillslope/Jane Atha, WDFW

Big Beef Creek, Kitsap Peninsula/Ned Pittman, WDFW

There is substantial agreement among scientists as to the potential magnitude and consequences of peak flow changes due to forestry. Grant et al. (2008) assessed the effects of forestry on peak flows and consequent channel response in the Pacific Northwest in a state of the science report. Their findings, primarily synthesized from long-term monitoring, show that responses are complex, and vary with runoff type (snowmelt, rainfall or rain-on-snow), local hydrology, harvest treatment, and channel type. The authors generally noted that geomorphic changes from harvest-related peak flow effects are limited to low-gradient channels (below 2% grade) and are generally minor as compared to other anthropomorphic impacts. Other literature reviews (e.g., Liquori et al. 2008; Grant et al. 1999) had similar conclusions.

The degree to which forestry practices alter water and sediment yields depends on where the logging occurs within a drainage basin. In one study, logging at lower elevations in snowmelt-dominated basins of southwestern Canada caused little to no change in peak flow because of relatively small snowpack, whereas in the higher elevations the peak flow changed significantly (Whitaker et al. 2002). Similarly, characteristics such as climate, topography, and soils also determine landscape sensitivity to forestry. Several studies have shown that mean annual precipitation is a reasonable indicator of impact (e.g. Anderson et al. 1976; Coe 2006; CBOF-TAC 2008). A road network may deliver 20% of sediment to stream channels in areas with 22 in (500 mm) per year average precipitation compared with 50% of the sediment when precipitation exceeds 118 in (3,000 mm) per year (Coe 2006). In areas of high topographic relief, it is particularly challenging for researchers to differentiate channel adjustments induced by logging from change that would have occurred without interference (Marston 2008).

Conversion of riparian areas from forest to other land uses can cause channel narrowing and deepening (Anderson et al. 2004; Sweeney et al. 2004; Faustini et al. 2009). Furthermore, streams without riparian forests often exhibit channels with less wood that are narrower, deeper, have fewer obstructions, less variability in active channel width, higher proportions of run and glide habitats, and lower habitat diversity (Jackson et al. 2014).

Road Infrastructure

Road networks are similar to stream networks in that they are often widely distributed, transport energy and material across a landscape, and have a high edge length per unit area enabling interaction with adjacent hillslopes (Jones et al. 2000). Stream networks and road networks commonly occur in similar densities in wet-climate forested mountains where logging has occurred (Wemple et al. 1996).

The biggest impact of unpaved roads on aquatic systems is sedimentation. Road construction and maintenance disturbs a layer of soil on the road tread, adjacent drainage ditch, and cutslopes that becomes easily eroded material (Megahan 1974). In conjunction with other loose material on roads, higher erodibility increases the total sediment yield from a road segment (Luce and Black 1999). In managed forests of Washington State, unpaved roads and trails are the most significant management activity that affects sediment production and delivery into streams (Lewis 1998; Gomi et al. 2005; CBOF TAC 2008). Roads can initiate mass movements that tend to result in longer runout zones and more sediment than their naturally occurring counterparts do (May 2002). Sediment generated from roads can be delivered to streams via ditches and gullies below cross-draining culverts that concentrate and route runoff. Road sediment production varies substantially with the type of surfacing material, road slope, mean annual precipitation, geology, traffic type and volumes, and road area (Cafferata and Munn 2002; MacDonald et al. 2004; Coe 2006). Roads and water crossings with improper design or maintenance are often the most significant sources of erosion and sediment delivery to streams in forested watersheds (Madej 2001; Cafferata and Munn 2002). Jones et al. (2000) found that unpaved roads continue to contribute sediment fines to channels long after harvested vegetation has reestablished.

Agriculture

Agriculture, in the form of livestock grazing or crop production, can reduce and alter vegetation cover, compact soils,and reduce water infiltration. It can also increase runoff and sediment yield, and cause associated changes in streamflow, channel morphology, and channel stability.

Cattle grazing can severely degrade riparian areas particularly through vegetation reduction and trampling of streambanks. Grazing animals can create ramps along streambanks and trails along the floodplain that enhance localized erosion (Trimble and Mendel 1995). Grazing-disturbed streams are usually characterized by a lack of overhanging banks, wide and shallow channels, smaller pool area, high sediment yields and turbidity, steep gradients, and channel beds typified by long stretches of glides or runs. Along streams with enclosures that kept cattle out of riparian areas, Magilligan and McDowell (1997) found that channel widths became narrower due to increased roughness by grassy riparian vegetation, which ultimately traps sediment leading to width reductions. The channel narrowing effects of grassy vegetation were also reported by Sweeney et al. (2004) and Allmendinger et al. (2005).

Understanding a channel's state of equilibrium (or disequilibrium) and its potential future states is critical for restoration and protection of salmon habitats.

A common effect of planting crops is increased sediment yield and associated changes in channel pattern and stability (Wohl 2014). The magnitude of these changes is dependent on the type and extent of crops, as well as the topographic and soil characteristics at a site. Numerous case studies indicate that soil erosion from conventional agriculture exceeds rates of natural erosion by up to several orders of magnitude (Montgomery 2007).

Urban/Suburban

Research on the effects of urbanization on sediment yield and channel response date back almost 50 years and is global in reach (Wolman 1967). The results of this research draw similar conclusions regarding the sequence of changes that occur within a drainage basin, with differences occurring only in the magnitude and timing (Chin 2006).

Erosion rates can reach up to 40,000 times pre-disturbance rates on land surfaces cleared for building that remain bare for up to a year (Harbor 1999). Construction sites can dramatically increase sediment yield from an urbanizing basin (Fusillo et al. 1977). For example, sediment production from Issaquah Creek

Mill Creek, Walla Walla, Washington/Jane Atha, WDFW

watershed in western Washington had two- to five-fold increases when only 0.3% of its 55.6 mi^2 (144 km^2) drainage area was under construction (Nelson and Booth 2002). Upon completion of construction, sediment yield usually decreases to negligible amounts as urbanizing areas are stabilized beneath roads, buildings, and lawns.

Research on the effects of urbanization on hydrology and flooding is also extensive, and dates back at least 50 years. Collectively, this research clearly shows that urban development, and associated increases in impervious surfaces, leads to larger and more frequent floods (Leopold 1968). The key metrics of change are peak discharge, lag time, flood frequency, and total runoff or water yield. Morphological adjustments have been studied in response to hydrologic changes, with most studies showing increases in channel width (Chin 2006).

Stream Channel Modifications

Humans have been directly altering stream flow and channel form for centuries (Poff et al. 1997). Flow regulation, through dams, reduces the size and variation of annual peak flows (Williams and Wolman 1984), increases minimum flows (Hirsch et al. 1990), shifts the timing of seasonal flows, and can alter diurnal flow fluctuations if the dam is used for hydroelectric power generation (Magilligan and Nislow 2001).

Dams can have dramatic effects on aquatic and riparian systems (Nilsson et al. 1997; Nilsson and Berggren 2000; Katz et al. 2005; Merritt and Wohl 2006). Dams disconnect the channel from the floodplain by reducing peak flows. This decreases or eliminates freshly scoured surfaces necessary for seedling establishment in riparian areas. Flow regulation can disrupt downstream transport of seeds to germination sites, and seedlings established on gravel bars may die from prolonged submersion when managed flows are higher than natural flows. Changes in sediment size distributions and moisture content can also adversely impact riparian vegetation. The cumulative effects on riparian communities downstream of dams can result in fewer plant species and lower plant density (Jansson et al. 2000).

Humans regularly alter channel form through dredging, channel straightening, removing instream wood or Beaver dams, building check dams, extending channel networks via canals, burying or laterally shifting inconvenient channels, and simply reconfiguring a channel to be more esthetically pleasing. (Wohl 2014). Levees are a direct form of channel alteration particularly ubiquitous in lowland rivers. Levees are mounds built parallel to stream channels that limit overbank flooding, and are also known as dikes or embankments (Petroski 2006). Levees severely reduce or eliminate channel-floodplain exchanges, facilitate higher magnitude floods, and exacerbate flooding in downstream areas without levees. Disconnection between the channel and floodplain reduces deposition of sediment and organic matter in floodplains, leading to loss of habitat, reduced animal abundance, and lower biodiversity in riparian areas (Hohensinner et al. 2004).

Lastly, humans can indirectly and severely alter channel form by introducing invasive, exotic species to riparian areas. Invasive plants may displace native riparian plants and thus impact streambank resistance to erosion and overbank sedimentation dynamics (Graf 1978; Allred and Schmidt 1999). Invasive riparian plants can also change patterns of water uptake and transpiration which in turn alter streamflows, subsurface water tables, nutrient cycling, and the quality and quantity of aquatic habitats (Schilling and Kiniry 2007; Hultine and Bush 2011).

2.6. Fish and Wildlife Habitat

2.6.1. Fish

The fisheries and fish ecology literature are replete with studies that report relationships between fish and physical habitat conditions. Such studies demonstrate the profound importance of instream conditions for fish and they have helped guide habitat management and restoration. For example, Nickelson et al. (1992) found seasonal differences in the use of pool and riffle habitat by juvenile Coho Salmon *Oncorhynchus kisutch* and suggested that additional pool habitats might improve their survival. Such correlative approaches have been expanded by considering shifts in habitat use within species and differential habitat requirements among species in attempts to identify optimal habitat type configurations (Rosenfeld 2003; Bain and Jai 2012). However, the success of such approaches has been limited by high spatiotemporal variability in, for example, habitat-density relations among streams (Dunham and Vinyard 1997; Dunham et al. 2002) that can be due to biotic and abiotic factors. Further, population statistics such as density can be a poor indicator of habitat quality (Van Horne 1983).

Juvenile Chinook Salmon Oncorhynchus tshawytscha/*David Price, WDFW*

Recent scientific advances regarding the protection and restoration of fish habitats (and habitats for other vagile aquatic species) include the following: 1) acknowledgement of the importance and frequency of fish movement (Schlosser 1991, 1995; Gowan et al. 1994), 2) greater acknowledgement of the importance of habitat heterogeneity and spatial variability in fish-habitat relationships (Torgersen et al. 2006), and 3) the concept of riverscapes (Ward 1998; Fausch et al. 2002; Allan 2004), which describes a dynamic mosaic of habitat types and environmental gradients that are characterized by high connectivity and complexity.

The riverscape concept expands on the classic stream continuum concept of Vannote et al. (1980) mostly by explicitly incorporating spatial and temporal heterogeneity, discontinuities, and connectivity between stream reaches (longitudinal), the stream and uplands (lateral), and the stream and groundwater (vertical). Fausch et al. (2002) provide several principles for effective research and management given the riverscape concept. These include: 1) conduct research (and management) at appropriate scales for the question, 2) the importance of physical and ecological processes are revealed at different spatiotemporal scales and processes will interact among scales, 3) rare or unique features can be very important, and 4) unintended consequences of habitat degradation propagate in all directions, including upstream. Cumulatively, these principles and the vast research that supported their development suggest that the spatial extent and temporal duration of research and management must be matched to the scales at which populations of species use habitat and at which suitable habitat conditions are created and maintained. Further, they emphasize the importance of maintaining and restoring longitudinal and lateral connectivity of stream systems (Sedell et al. 1989) to allow for proper functioning of habitat forming processes and maintenance and restoration of supplementary and complimentary habitats to account for expected variability in local suitability.

Conducting research and management that incorporates the riverscape concept is challenging because it often requires study at large spatial and long temporal extents as well as monitoring biotic and abiotic processes at fine spatial and temporal resolutions. Perhaps the best examples of such work in Washington are the Intensively Monitored Watersheds (IMW) projects that are designed to assess the efficacy of stream restoration for increasing the freshwater survival and production of salmon (Bilby et al. 2005). The IMW projects are conducted at the spatial extent of watersheds and over the course of several salmon life cycles. Important physical and ecological processes are monitored within and among individual stream reaches with the intent of identifying rare features and measuring changes in habitat complexity and connectivity (Bennett et al. 2016). This approach should yield reliable inferences regarding the efficacy of restoration efforts that can be used to inform decisions via adaptive management (Bennett et al. 2016).

2.6.2. Amphibians and Reptiles

Amphibians are less conspicuous inhabitants of instream habitats than fish, but in much of the Pacific Northwest, particular groups of stream-breeding amphibian species that include giant salamanders (*Dicamptodon* spp.), tailed frogs (*Ascaphus* spp.), and torrent salamanders (*Rhyacotriton* spp.) dominate fishless headwater

The riverscape concept expands on Vannote et al.'s (1980) classic stream continuum concept by including lateral and vertical connectivity and incorporating spatial and temporal heterogeneity.

Coastal Tailed Frog Ascaphus truei *in metamorphosis/Marc Hayes, WDFW*

streams. Headwater streams represent a large majority (up to 80%) of the overall stream network based on its length (Meyer and Wallace 2001). Stream-breeding amphibian dominance in headwaters partly reflects the flow conditions in headwater habitats and the suitable physical conditions those flows create. Amphibian abundance is highest in step-pool and cascade reach morphologies characterized by armored beds, stable bedforms, refuge spaces amongst coarse bed material, and reduced tractive forces as a result of tumbling flow (Dupuis and Friele 2006). For example, torrent salamanders, the only stream-breeding amphibian group that does not attach their eggs to rocky substrates, deposit and conceal them in the lowest flow headwater ends of streams and their tributaries; higher flow habitats will dislodge or damage their eggs. In contrast, giant salamanders and tailed frogs, which glue their eggs to coarse rocky substrates, use concealed oviposition sites in stable step structures typically found further downstream in the headwater network (Hayes et al. 2006). Stream-breeding amphibians seem to become less abundant below headwater streams at least in part because substrate and flow conditions that provide either oviposition or refuge sites become less frequent (Brummer and Montgomery 2003). Other conditions, such as predation by fishes, also play a role.

The freshets (high water events) in larger streams or rivers that have an alluvial floodplain, when coupled with wood, are also critical to both creation and maintenance of off-channel amphibian habitats (Amoros and Bornette 2002). Off-channel habitats comprise among the most important lowland stillwater habitats representing key breeding and rearing areas for the largest group of native amphibians (stillwater breeders) (Richter 2005) and the garter snakes (*Thamnophis* spp.) that feed on them. Such off-channel habitats are also critical rearing and foraging habitat for native turtles (Holland 1994).

2.7. Conclusions

The interactions of water, sediment, wood, and riparian vegetation create complex channel morphologies and diverse aquatic habitat conditions. Reach-scale differences in sediment erosion, transport, and deposition result in a mosaic of aquatic habitat conditions within and among stream reaches.

Understanding a channel's state of equilibrium (or disequilibrium) and its potential future states is critical for restoration and protection of salmon habitats. Consistent with the idea of process domains, not all stream reaches within a watershed are equally sensitive or resilient to human land uses or other anthropogenic disturbances. Describing a channel's historical range of natural variability can aid in understanding a reach's potential response to human disturbance. A historical perspective may also aid in identifying biological communities that are in most need of habitat restoration (McDonald et al. 2004; Brierley and Fryirs 2009). By adopting a historical perspective, we may be able to move closer to the ecosystem composition and structure, and the resulting habitat heterogeneity, essential for fish and wildlife.

Historically, disturbance and successional processes were key drivers for the creation of aquatic habitats (Reeves et al. 1995). Now and into the future, both natural and anthropogenic disturbances will affect the creation, maintenance, destruction, and recreation of aquatic habitats. Management may most reliably result in high quality habitat when it emulates historical disturbances and their effects on a watershed (Poff et al. 1997). An idea offered by Naiman et al. (2000) is to base management on historical patterns of watershed conditions and natural disturbances. If flooding was a major natural disturbance in a watershed, then with empirical data and/or models on the extent to which flooding has affected the watershed, one could describe the dynamic equilibrium conditions of channels and floodplains over time, and then tailor riparian management to the unique conditions of that watershed. If wildfires followed by landslides were the historical disturbance pattern in a watershed, then watershed-scale, long-term forest management plans might schedule timber harvests such that historical natural patterns are partially emulated (Reeves et al. 1995; Bisson et al. 2009). Emulating flood or wildlife disturbances for the benefit of salmon and other aquatic species will be a much more complicated project in residential (i.e., rural, suburban, urban) areas.

The importance of riparian ecosystems in maintaining channel, off-channel, and floodplain habitats is increasingly well supported by the scientific literature. In particular, the importance of longitudinal, lateral, and vertical connectivity to the quality of habitat conditions and the importance of riparian vegetation to stream channel stability are increasingly emphasized. Management of riparian ecosystems should consider connectivity among the geomorphic process domains, the influences of natural disturbance, the historical range of natural variability, and the inextricable influence that riparian vegetation has on geomorphic processes.

Bridge on Strait of Juan de Fuca Highway/Ned Pittman, WDFW

2.8. Literature Cited[5]

Agee, J.K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, D.C. (i)

Allan, J.D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 35:257-284. (i)

Allen, J.R.L. 1965. A review of the origin and characteristics of recent alluvial sediments. Sedimentology 5:89-191. (i)

Allmendinger, N.E., J.E. Pizzuto, N. Potter, T.E. Johnson, and W.C. Hession. 2005. The influence of riparian vegetation on stream width, eastern Pennsylvania, USA. Geological Society of America Bulletin 117:229-243. (i)

Allred, T.M., and J.C. Schmidt. 1999. Channel narrowing by vertical accretion along the Green River near Green River, Utah. Geological Society of America Bulletin 111:1757-1772. (i)

Amoros, C., and G. Bornette. 2002. Connectivity and biocomplexity in waterbodies of riverine floodplains. Freshwater Biology 47:761-776. (i)

Anderson, J.R., E.E. Hardy, J.T. Roach, and R.E. Witmer. 1976. A land use and land cover classification system for use with remote sensor data. Geological Survey Professional Paper 964. U.S. Geological Survey, Arlington, Virginia. (viii)

Anderson, R.J., B.P. Bledsoe, and W.C. Hession. 2004. Widths of streams and rivers in response to vegetation, bank material, and other factors. Journal of the American Water Resources Association 40:1159-1172. (i)

ASCE (American Society of Civil Engineers Task Committee on Hydraulics, Bank Mechanics, and Modelling of River Width Adjustment). 1998. River width adjustment. I: processes and mechanisms. Journal of Hydraulic Engineering 124:881-902. (i)

Bain, M.B., and H. Jai. 2012. A habitat model for fish communities in large streams and small rivers. International Journal of Ecology 2012:962071. (i)

Beechie, T.J., M. Liermann, M.M. Pollack, S. Baker, and J. Davies. 2006. Channel pattern and river-floodplain dynamics in forested mountains river systems. Geomorphology 78:124-141. (i)

Benda, L., M.H. Hassan, M. Church, and C.L. May. 2005. Geomorphology of steepland headwaters: the transition from hillslopes to channels. Journal American Water Resources Association 41:835-851. (i)

Bennett, S., G. Pess, N. Bouwes, P. Roni, R.E. Bilby, S. Gallagher, J. Ruzycki, T. Buehrens, K. Krueger, W. Ehinger, J. Anderson, C. Jordan, B. Bowersox, and C. Greene. 2016. Progress and challenges of testing the effectiveness of stream restoration in the Pacific Northwest using intensively monitored watersheds. Fisheries 41:92-103. (i)

Betson, R.P. 1964. What is watershed runoff? Journal of Geophysical Research 69:1541–1552. (i)

Bisson, P.A., J.B. Dunham, and G.H. Reeves. 2009. Freshwater ecosystems and resilience of Pacific salmon: habitat management based on natural variability. Ecology and Society 14(1):45. (i)

Bilby, R., W. Ehinger, T. Quinn, G. Volkhardt, K. Krueger, D. Seiler, G. Pess, C. Jordan, M. McHenry, and D. Poon. 2005. Study evaluates fish response to management actions. Western Forester 50:14-15. (i)

Boulton, A.J., S. Findlay, P. Marmonier, E.H. Stanley, and H.M. Valett. 1998. The functional significance of the hyporheic zone in streams and rivers. Annual Review Ecology and Systematics 29:59-81. (i)

Brierley, G., and K. Fryirs. 2009. Don't fight the site: three geomorphic considerations in catchment-scale river rehabilitation planning. Environmental Management 43:1201-1218. (i)

Brummer, C.J., and D.R. Montgomery. 2003. Downstream coarsening in headwater channels. Water Resources Research 39:1-13. (i)

Brunsden, D., and J.B. Thornes. 1979. Landscape sensitivity and change. Transactions of the Institute of British Geographers 4:463-484. (i)

Bryan, R. 1990. Knickpoint evolution in rillwash. Catena Supplement 17:111-132. (i)

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[5] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Burckhardt, J.C., and B.L. Todd. 1998. Riparian forest effect on lateral stream channel migration in the glacial till plains. Journal of the American Water Resources Association 34:179-184. (i)

Butler, B. 2004. Channel migration zones. Chapter 4 *in* Best available science, volume 1: a review of science literature. King County, Seattle. (viii)

Cafferata, P.H., and J.R. Munn. 2002. Hillslope monitoring program: monitoring results from 1996 through 2001. Monitoring Study Group Final Report prepared for the California State Board of Forestry and Fire Protection. Sacramento, California. (viii)

Callander, R.A. 1978. River meandering. Annual Review of Fluid Mechanics 10:129-158. (i)

CBOF-TAC (California Board of Forestry and Fire Protection Technical Advisory Committee). 2008. Primer on water riparian exchanges related to forest management in the western U.S., Version 2.0. Sacramento, California. (viii)

Chin, A. 2006. Urban transformation of river landscapes in a global context. Geomorphology 79:460-487. (i)

Coe, D. 2006. Sediment production and delivery from forest roads in the Sierra Nevada, California. Master's thesis. Colorado State University, Fort Collins. (i)

Collins, B.F., and D.R. Montgomery. 2002. Forest development, wood jams, and restoration of floodplain rivers in the Puget Lowland, Washington. Restoration Ecology 10:237-247. (i)

Corenblit, D., E. Tabacchi, J. Steiger, and A.M. Gurnell. 2007. Reciprocal interactions and adjustments between fluvial landforms and vegetation dynamics in river corridors: a review of complementary approaches. Earth-Science Reviews 84:56-86. (i)

Dietrich, W., and T. Dunne. 1978. Sediment budget for a small catchment in a mountainous terrain. Pages 31- 48 *in* O. Slaymaker, editor. Fluvial geomorphology. Rutledge, New York. (i)

Dietrich, W.E., J.D. Smith, and T. Dunne. 1984. Boundary shear stress, sediment transport, and bed morphology in a sand-bedded river meander during high and low flow. Pages 632-639 *in* C.M. Elliot, editor. River meandering: proceedings of the conference rivers '83. American Society of Civil Engineers, New York. (viii)

Dunham, J.B., B.S. Cade, and T.W. Terrell. 2002. Influences of spatial and temporal variation on fish-habitat relationships defined by regression quantiles. Transactions of the American Fisheries Society 131:86-98. (i)

Dunham, J.B., and G.L. Vinyard. 1997. Incorporating stream level variability into analyses of site level fish habitat relationships: some cautionary examples. Transactions of the American Fisheries Society 126:323-329. (i)

Dunne, T. 1978. Field studies of hillslope flow processes. Pages 227-293 *in* M.J. Kirby, editor. Hillslope hydrology. John Wiley, Chichester, United Kingdom. (i)

Dunne, T., and R.D. Black. 1970a. Partial-area contributions to storm runoff in a small New England watershed. Water Resources Research 6:1296-1311. (i)

Dunne, T., and R.D. Black. 1970b. An experimental investigation of runoff production in permeable soils. Water Resources Research 6:170-191. (i)

Dunne, T., and L.B. Leopold. 1978. Water in environmental planning. WH Freeman and Company, San Francisco. (i)

Dunne, T., D.V. Malmon, and S.M. Mudd. 2010. A rain splash transport equation assimilating field and laboratory measurements. Journal of Geophysical Research 115:F01001. (i)

Dupuis, L, and P. Friele. 2006. The distribution of the Rocky Mountain Tailed Frog (*Ascaphus montanus*) in relation to the fluvial system: implications for management and conservation. Ecological Research 21:489-502. (i)

Dust, D., and E. Wohl. 2012. Conceptual model for complex river responses using an expanded Lane's relation. Geomorphology 139:109-121. (i)

Eis, S. 1987. Root systems of older immature hemlock, cedar and Douglas-fir. Canadian Journal of Forest Research 17:1348-1354. (i)

Everett, R., R. Schellhaas, P. Ohlson, D. Spurbeckand, and D. Keenum. 2003. Continuity in fire disturbance between riparian and adjacent sideslope Douglas-fir forests. Forest Ecology and Management 175:31-47. (i)

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management: an ecological, economic and social assessment. U.S. Department of Agriculture and U.S. Department of the Interior, Portland, Oregon. (viii)

Fausch, K.D., C.E. Torgersen, C.V. Baxter, and H.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52:1-16. (i)

Faustini, J.M., Kaufmann, P.R., and A.T. Herlihy. 2009. Downstream variation in bankfull width of wadeable streams across the conterminous United States. Geomorphology 108:292-311. (i)

Florsheim, J.L., J.F. Mount, and A. Chin. 2008. Bank erosion as a desirable attribute of rivers. BioScience 58:519-529. (i)

Foley, J.A., R. DeFries, G.P. Asner, C. Barford, G. Bonan, S.R. Carpenter, F.S. Chapin, M.T. Coe, G.C. Daily, and H.K. Gibbs. 2005. Global consequences of land use. Science 309:570-574. (i)

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream classification: Viewing streams in a watershed context. Environmental Management 10:199-214. (i)

Furbish, D.J., E.M. Childs, P.K. Haff, and M.W. Schmeeckle. 2009. Rain splash of soil grains as a stochastic advection dispersion process, with implications for desert plant soil interactions and land surface evolution. Journal of Geophysical Research 114:F00A03. (i)

Fusillo, T.V., G. Nieswand, and T. Shelton. 1977. Sediment yields in a small watershed under suburban development. Pages 34-88 *in* Proceedings from the international symposium on urban hydrology, hydraulics, and sediment control. University of Kentucky Press, Lexington. (viii)

Gilbert, G.K. 1877. Report on the geology of the Henry Mountains. U.S. Government Printing Office, Washington, D.C. (viii)

Gomi, T., R.D. Moore, and M. Hassan. 2005. Suspended sediment dynamics in small forest streams of the Pacific Northwest. Journal of the American Water Resources Association 41:877-898. (i)

Gorrick, S., and J.F. Rodrígue. 2012. Sediment dynamics in a sand bed stream with riparian vegetation. Water Resources Research 48:W02505. (i)

Gowan, C., M.K. Young, K.D. Fausch, and S.C. Riley. 1994. Restricted movement of resident stream salmonids: a paradigm lost. Canadian Journal of Fisheries and Aquatic Sciences 51:2626-2637. (i)

Graf, W.L. 1978. Fluvial adjustments to the spread of tamarisk in the Colorado Plateau region. Geological Society of America Bulletin 89:1491-1501. (i)

Grant, G.E., S.L. Lewis, F.J. Swanson, J.H. Cissel, and J.J. McDonnell. 2008. Effects of forest practices on peak flows and consequent channel response: a state-of-the-science report for western Oregon and Washington. PNW-GTR-760. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Grant, G., W. Megahan, and R. Thomas. 1999. A re-evaluation of peak flows: do forest roads and harvesting cause floods? Paper presented at the 1999 NCASI west coast regional meeting. National Council for Air and Stream Improvement, Corvallis, Oregon. (viii)

Grant G.E., J.E. O'Connor J.E., and M.G. Wolman. 2013. A river runs through it: conceptual models in fluvial geomorphology. Pages 6-21 *in* J.F. Shroder, editor. Treatise on geomorphology, volume 9. Academic Press, San Diego, California. (i)

Griffin, E., J. Kean, K. Vincent, J. Smith, and J. Friedman. 2005. Modeling effects of bank friction and woody bank vegetation on channel flow and boundary shear stress in the Rio Puerco, New Mexico. Journal of Geophysical Research: Earth Surface 110:F04023. (i)

Güneralp, I., and R.A. Marston. 2012. Process-form linkages in meander morphodynamics: bridging theoretical modeling and real world complexity. Progress in Physical Geography 36:718-746. (i)

Gurnell, A. 2014. Plants as river system engineers. Earth Surface Processes and Landforms 39:4-25. (i)

Hairston-Strang, A.B., and P.W. Adams. 2000. Riparian management area condition for timber harvests conducted before and after the 1994 Oregon water protection rules. Western Journal of Applied Forestry 15:147-153. (i)

Hales, T., and J. Roering. 2009. A frost "buzzsaw" mechanism for erosion of the eastern Southern Alps, New Zealand. Geomorphology 107:241-253. (i)

Harbor, J. 1999. Engineering geomorphology at the cutting edge of land disturbance: erosion and sediment control on construction sites. Geomorphology 31:247-263. (i)

Harris, D.D. 1977. Hydrologic changes after logging in two small Oregon coastal watersheds. Water Supply Paper 2037. U.S. Geological Survey, Washington, D.C. (i)

Hayes, M., T. Quinn, D. Dugger, T. Hicks, M. Melchiors, and D. Runde. 2006. Dispersion of Coastal Tailed Frog (*Ascaphus truei*): a hypothesis relating occurrence of frogs in non-fish-bearing headwater basins to their seasonal movements. Journal of Herpetology 40:533-545. (i)

Heede, B.H. 1980. Stream dynamics: an overview for land managers. PNW-GTR-72. U.S. Forest Service. Rocky Mountain Research Station, Fort Collins, Colorado. (viii)

Henning, J.A., and G. Schirato. 2006. Amphibian use of Chehalis River floodplain wetlands. Northwestern Naturalist 87:209-214. (i)

Hewlett, J.D., and W.L. Nutter. 1970. The varying source area of streamflow from upland basins. Pages 65-83 *in* Proceedings of the symposium on interdisciplinary aspects of watershed management. American Society of Civil Engineers, New York. (viii)

Hickin, E.J. 1984. Vegetation and river channel dynamics. Canadian Geographer 28:111-126. (i)

Hirsch, R., J. Walker, J. Day, and R. Kallio. 1990. The influence of man on hydrologic systems. Pages 329–359 *in* M.G. Wolman and H.C. Riggs, editors. Surface water hydrology. Geological Society of America, Boulder, Colorado. (viii)

Hohensinner, S., H. Habersack, M. Jungwirth, and G. Zauner. 2004. Reconstruction of the characteristics of a natural alluvial river-floodplain system and hydromorphological changes following human modifications: the Danube River (1812-1991). River Research and Applications 20:25-41. (i)

Holland, D.C. 1994. The Western Pond Turtle: habitat and history. Final Report to the U.S. Department of Energy, Bonneville Power Administration, Portland, Oregon. (viii)

Hooke, J.M. 1984. Changes in river meanders: a review of techniques and results of analyses. Progress in Physical Geography 8:473-508. (i)

Horton, R.E. 1933. The role of infiltration in the hydrologic cycle. Transactions American Geophysical Union, 14:446-460. (i)

Hultine, K., and S. Bush. 2011. Ecohydrological consequences of non-native riparian vegetation in the southwestern United States: a review from an ecophysiological perspective. Water Resources Research 47:W07542. (i)

Hughes, F.M.R. 1997. Floodplain biogeomorphology. Progress in physical geography 21:501–529. (i)

Hupp, C.R., and W.R. Osterkamp. 1996. Riparian vegetation and fluvial geomorphic processes. Geomorphology 14:277-295. (i)

Jackson, C., D. Leigh, S. Scarbrough, and J. Chamblee. 2014. Herbaceous versus forested riparian vegetation: narrow and simple versus wide, woody and diverse stream habitat. River Research and Applications 31:847-857. (i)

Jansson, R., C. Nilsson, M. Dynesius, and E. Andersson. 2000. Effects of river regulation on river-margin vegetation: a comparison of eight boreal rivers. Ecological applications 10:203-224. (i)

Jones, J.A., F.J. Swanson, B.C. Wemple, and K.U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conservation Biology 14:76-85. (i)

Katz, G.L., J.M. Friedman, and S.W. Beatty. 2005. Delayed effects of flood control on a flood-dependent riparian forest. Ecological Applications 15:1019-1035. (i)

Kauffman, J.B., and R.E. Martin. 1989. Fire behavior, fuel consumption, and forest floor changes following prescribed understory fires in Sierra Nevada mixed conifer forests. Canadian Journal of Forest Resources 19:455-462. (i)

Kiffney, P.M., C.M. Greene, J.E. Hall, and J.R. Davies. 2006. Tributary streams create spatial discontinuities in habitat, biological productivity and diversity in mainstem rivers. Canadian Journal of Fisheries and Aquatic Sciences 63:2518-2530. (i)

Knighton, D. 2014. Fluvial forms and processes: a new perspective. Routledge, London. (i)

Krause, D., D.M. Hannah, J.H. Fleckenstein, C.M. Heppell, D Kaeser, R. Pickup, G. Pinay, A.L. Robertson, and P.J. Wood. 2011. Inter-disciplinary perspectives on processes in the hyporheic zone. Ecohydrology 4:481-499. (i)

Kupfer, J., and G.P. Malanson. 1993. Observed and modeled directional change in riparian forest composition at a cutbank edge. Landscape Ecology 8:185-199. (i)

Lane, E.W. 1955. The importance of fluvial morphology in hydraulic engineering. Proceedings of American Society of Civil Engineers 81:1-17. (i)

Langendoen, E.J., R.R. Lowrance, and A. Simon. 2009. Assessing the impact of riparian processes on streambank stability. Ecohydrology 2:360-369. (i)

Leopold, L.B. 1968. Hydrology for urban land planning: A guidebook on the hydrologic effects of urban land use. Geological Survey Circular 554. U.S. Geological Survey, Reston, Virginia. (i)

Leopold, L.B., and G. Wolman. 1957. River channel patterns: braided, meandering and straight. U.S. Geological Survey Professional Paper 282-B. U.S. Government Printing Office, Washington, D.C. (i)

Lewis, J. 1998. Evaluating the impacts of logging activities on erosion and sediment transport in the Caspar Creek watersheds. Pages 55-69 *in* R.R. Ziemer, technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story. PSW-GTR-168. U.S. Forest Service, Pacific Southwest Research Station, Albany, California. (viii)

Liquori, M.K. 2006. Post-harvest riparian buffer response: implications for wood recruitment modeling and buffer design. Journal of American Water Resources Association 42(1):177-189. (i)

Liquori, M.K., D. Martin, R. Coats, L. Benda, and D. Ganz. 2008. Scientific literature review of forest management effects on riparian functions for anadromous salmonids. Sound Watershed Consulting, Oakland, California. (viii)

Luce, C.H., and T.A. Black. 1999. Sediment production from forest roads in western Oregon. Water Resources Research 35:2561-2570. (i)

MacDonald, L.H., D.B. Coe, and S.E. Litschert. 2004. Assessing cumulative watershed effects in the central Sierra Nevada: hillslope measurements and catchment-scale modeling. Pages 149-157 *in* D.D. Murphy and P.A. Stine, editors. Proceedings of the Sierra Nevada science symposium. PSW-GTR-193. U.S. Forest Service, Pacific Southwest Research Station, Albany, California. (viii)

Madej, M.A. 2001. Erosion and sediment delivery following removal of forest roads. Earth Surface Processes and Landforms 26:175-190. (i)

Madej, M.A., and V. Ozaki. 1996. Channel response to sediment wave propagation and movement, Redwood Creek, California, USA. Earth Surface Processes and Landforms 21:911-927. (i)

Magilligan, F.J., and P.F. McDowell. 1997. Stream channel adjustments following elimination of cattle grazing. Journal of the American Water Resources Association 33:867-878. (i)

Magilligan, F.J., and K.H. Nislow. 2001. Long-term changes in regional hydrologic regime following impoundment in a humid-climate watershed. Journal of the American Water Resources Association 37:1551-1569. (i)

Mahoney, J.M., and S.B. Rood. 1998. Streamflow requirements for cottonwood seedling recruitment -an integrative model. Wetlands 18:634-645. (i)

Malanson, G.P. 1993. Riparian Landscapes. Cambridge University Press, Cambridge, United Kingdom. (i)

Marston, R.A. 2008. Land, life, and environmental change in mountains. Annals of the Association of American Geographers 98:507-520. (i)

Mattson, W.J., and N.D. Addy. 1975. Phytophagous, insects as regulators of forest primary production. Science 190:515-522. (i)

May, C.L. 2002. Debris flows through different forest age classes in the central Oregon Coast Range, Journal of the American Water Resources Association 38:1097-1113. (i)

McDonald, A., S.N. Lane, N.E. Haycock, and E. Chalk. 2004. Rivers of dreams: on the gulf between theoretical and practical aspects of an upland river restoration. Transactions of the Institute of British Geographers 29:257-281. (i)

McDonnell, J.J. 2003. Where does water go when it rains? Moving beyond the variable source area concept of rainfall-runoff response. Hydrological Processes 17:1869-1875. (i)

Megahan, W.F. 1974. Erosion over time on severely disturbed granitic soils: a model. INT-156. U.S. Forest Service, Intermountain Forest & Range Experiment Station, Ogden, Utah. (i)

Meitzen, K.M. 2009. Lateral change migration effects on riparian forest structure and composition, Congaree River, South Carolina, USA. Wetlands 29:465-475. (i)

Merritt, D.M., and E.E. Wohl. 2006. Plant dispersal along rivers fragmented by dams. River Research and Applications 22:1-26. (i)

Meyer, L., G. Foster, and S. Nikolov. 1975. Effect of flow rate and canopy on rill erosion. Transactions of the American Society of Agricultural Engineers 18:905-911. (i)

Meyer, J.L., and J.B. Wallace. 2001. Lost linkages and lotic ecology: rediscovering small streams. Pages 295-317 *in* M.C. Press, N.J. Huntly, and S. Levin, editors. Ecology: achievement and challenge. Blackwell Science, Oxford, United Kingdom. (i)

Montgomery, D.R. 1999. Process domains and the river continuum. Journal of the American Water Resources Association 35:397-410. (i)

Montgomery, D.R. 2007. Is agriculture eroding civilization's foundation? GSA Today 17(10):4-9. (i)

Montgomery, D.R., T.B. Abbe, J.M. Buffington, N.P. Peterson, K.M. Schmidt, and J.D. Stock. 1996. Distribution of bedrock and alluvial channels in forested mountain drainage basins. Nature 381:587-589. (i)

Montgomery, D.R., and J.M. Buffington. 1997. Channel-reach morphology in mountain drainage basins. Geological Society of America Bulletin 109:596-611. (i)

Montgomery, D.R., K.M. Schmidt, W.E. Dietrich, and J. McKean. 2009. Instrumental record of debris flow initiation during natural rainfall: implications for modeling slope stability. Journal of Geophysical Research 114:F01031. (i)

Moore, R.D., and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review. Journal of the American Water Resources Association 41:763-784. (i)

Naiman, R.J., and R.E. Bilby. 1998. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Naiman, R.J., R.E. Bilby, and P.A. Bisson. 2000. Riparian ecology and management in the Pacific coastal rain forest. BioScience 50:996-1011. (i)

Naiman, R.J., H. Decamps, and M.E. McClain. 2010. Riparia: ecology, conservation, and management of streamside communities. Academic Press, London. (i)

Naiman, R.J., H. Decamps, and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3:209-212. (i)

Nakamura, F., and F.J. Swanson. 2003. Dynamics of wood in rivers in the context of ecological disturbance. Pages 279-298 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland. (i)

NRC (National Research Council). 2002. Riparian areas: strategies for management. National Academies Press, Washington, D.C. (i)

Nelson, E.J., and D.B. Booth. 2002. Sediment sources in an urbanizing, mixed land-use watershed. Journal of Hydrology 264:51-68. (i)

Nickelson, T.E., J.D. Rodgers, S.L. Johnson, and M.F. Solazzi. 1992. Seasonal changes in habitat use by juvenile Coho Salmon (*Oncorhynchus kisutch*) in Oregon coastal streams. Canadian Journal of Fisheries and Aquatic Sciences 49:783-789. (i)

Nilsson, C., and K. Berggren. 2000. Alterations of riparian ecosystems caused by river regulation. BioScience 50:783-792. (i)

Nilsson, C., R. Jansson, and U. Zinko, U., 1997. Long-term responses of river-margin vegetation to water-level regulation. Science 276:798-800. (i)

Nilsson, C., and M. Svedmark. 2002. Basic principles and ecological consequences of changing water regimes: riparian plant communities. Environmental Management 30:468-80. (i)

O'Connor, J.E., M.A. Jones, and T.L. Haluska. 2003. Floodplain and channel dynamics of the Quinault and Queets Rivers, Washington, USA. Geomorphology 51:31-59. (i)

Osterkamp, W.R., and C.R. Hupp. 2010. Fluvial processes and vegetation – glimpses of the past, the present, and perhaps the future. Geomorphology 116:274-285. (i)

Petroski, H. 2006. Levees and other raised ground. American Scientist 94(1):7-11. (i)

Pickett, S., S. Collins, and J. Armesto. 1987. A hierarchical consideration of causes and mechanisms of succession. Vegetatio 69:109-114. (i)

Pickett, S.T., and P.S. White. 1985. The ecology of natural disturbance and patch dynamics. Academic Press, San Diego, California. (i)

Pike, A.S., and F.N. Scatena. 2010. Riparian indicators of flow frequency in a tropical montane stream network. Journal of Hydrology 382:72-87. (i)

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B. Richter, R. Sparks, and J. Stromberg. 1997. The natural flow regime: a new paradigm for riverine conservation and restoration. BioScience 47:769-784. (i)

Pollen, N., and A. Simon. 2005. Estimating the mechanical effects of riparian vegetation on stream bank stability using a fiber bundle model. Water Resources Research 41:W07025. (i)

Pollen-Bankhead, N., and A. Simon. 2010. Hydrologic and hydraulic effects of riparian root networks on streambank stability: is mechanical root-reinforcement the whole story? Geomorphology 116:353–362. (i)

Polvi, L.E., E.E. Wohl, and D.M. Merritt. 2011. Geomorphic and process domain controls on riparian zones in the Colorado Front Range. Geomorphology 125:504-516. (i)

Polvi, L.E., E.E. Wohl, and D.M. Merritt. 2014. Modeling the functional influence of vegetation type on streambank cohesion. Earth Surface Processes and Landforms 39:1245-1258. (i)

Post, D.M., M.W. Doyle, J.L. Sabo, and J.C. Finlay. 2007. The problem of boundaries in defining ecosystems: a potential landmine for uniting geomorphology and ecology. Geomorphology 89:111-126. (i)

Rapp, C.F., and T.B. Abbe. 2003. A framework for delineating channel migration zones. Ecology Publication #03-06-027. Washington State Department of Ecology and Washington State Department of Transportation, Olympia. (viii)

Reeves, G.H., L.E. Benda, K.M. Burnett, P.A. Bisson, and J.R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. American Fisheries Society Symposium 17:334-349. (i)

Reid, L.M., and S. Hilton. 1998. Buffering the buffer. Pages 71-80 *in* R.R. Ziemer, technical coordinator. Proceedings of the conference on coastal watersheds: the Caspar Creek story. PSW-GTR-168. U.S. Forest Service, Pacific Southwest Research Station, Albany, California. (i)

Renschler, C.S., M.W. Doyle, and M. Thomas. 2007. Geomorphology and ecosystems: challenges and keys for success in bridging disciplines. Geomorphology 89:1-8. (i)

Richter, K.O. 2005. Wetlands. Pages 22-23 *in* L.L.C. Jones, W.P. Leonard, and D.J. Olson, editors. Amphibians of the Pacific Northwest. Seattle, Audubon Society, Seattle. (vi)

Rinaldi, M., N. Casagli, S. Dapporto, and A. Gargini. 2004. Monitoring and modelling of pore water pressure changes and riverbank stability during flow events. Earth Surface Processes and Landforms 29:237-254. (i)

Ritter, D.F., R.C. Kochel, and J.R. Miller. 2002. Process geomorphology. McGraw-Hill, Boston. (i)

Roering, J.J., K.M. Schmidt, J.D. Stock, W.E. Dietrich, and D.R. Montgomery. 2003. Shallow landsliding, root reinforcement, and the spatial distribution of trees in the Oregon Coast Range. Canadian Geotechnical Journal 40:237-253. (i)

Rosenfeld, J. 2003. Assessing the habitat requirements of stream fishes: an overview and evaluation of different approaches. Transactions of the American Fisheries Society 132:953-968. (i)

Rosenfeld J.S., E. Raeburn, P.C. Carrier, and R. Johnson. 2008. Effects of side channel structure on productivity of floodplain habitats for juvenile Coho Salmon. North American Journal of Fisheries Management 28: 1108–1119. (i)

Scanlon, B.R., I. Jolly, M. Sophocleous, and L. Zhang. 2007. Global impacts of conversions from natural to agricultural ecosystems on water resources: quantity versus quality. Water Resources Research 43:W03437. (i)

Schilling, K., and J. Kiniry. 2007. Estimation of evapotranspiration by Reed Canarygrass using field observations and model simulations. Journal of Hydrology 337:356-363. (i)

Schlosser, I.J. 1991. Stream fish ecology: a landscape perspective. BioScience 41:704-712. (i)

Schlosser, I.J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303:71-81. (i)

Schumm, S.A. 1977. The fluvial system. Wiley, New York. (i)

Schumm, S.A. 1979. Geomorphic thresholds: the concept and its applications. Transactions of the Institute of British Geographers 4:485-515. (i)

Sedell, J.R., J.E. Richey, and F.J. Swanson. 1989. The river continuum concept: a basis for the expected ecosystem behavior of very large rivers? Pages 49-55 *in* D.P. Dodge, editor. Proceedings of the international large river symposium. Department of Fisheries and Oceans, Ottawa, Canada. (i)

Selby, M.J. 1993. Hillslope forms and processes, 2nd edition. Oxford University Press, United Kingdom. (i)

Shankman, D. 1993. Channel migration and vegetation patterns in the Southeastern Coastal Plain. Conservation Biology 7:176-183. (i)

Simon, A., and A.J.C. Collison. 2002. Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. Earth Surface Processes and Landforms 27:527-546. (i)

Simon, A., A. Curini, S.E. Darby, and E.J. Langendoen. 2000. Bank and near-bank processes in an incised channel. Geomorphology 35:193-217. (i)

Simon, A., N. Pollen, and E. Langendoen. 2006. Influence of two woody riparian species on critical conditions for streambank stability: upper Truckee River, California. Journal of the American Water Resources Association 42:99-113. (i)

Sklar, L., and W.E. Dietrich. 1998. River longitudinal profiles and bedrock incision models: stream power and the influence of sediment supply. Geophysical Monograph Series 107:237-260. (i)

Smith, J.H.G. 1964. Root spread can be estimated from crown width of Douglas-fir, Lodgepole Pine, and other British Columbia tree species. Forestry Chronicle 40:456-473. (i)

Smith, B., P. Warkeand, and W. Whalley. 2002. Landscape development, collective amnesia and the need for integration in geomorphological research. Area 34:409-418. (i)

Smock, L.A., J.E. Gladden, J.L. Riekenberg, L.C. Smith, and C.R. Black. 1992. Lotic macroinvertebrate production in three dimensions: channel surface, hyporheic, and floodplain environments. Ecology 73:876-886. (i)

Stallins, J.A. 2006. Geomorphology and ecology: unifying themes for complex systems in biogeomorphology. Geomorphology 77:207-216. (i)

Stanford, J.A., and J.V. Ward. 1988. The hyporheic habitat of river ecosystems. Nature 335:64-66. (i)

Stanford, J.A., and J.V. Ward. 2001. Revisiting the serial-discontinuity concept. Regulated Rivers: Research & Management 17:303-310. (i)

Swanson, F.J., R.L. Fredriksen, and R.M. McCorison. 1982. Material transfer in a western Oregon forested watershed. Pages 233-266 *in* R.L. Edmonds, editor. Analysis of coniferous forest ecosystems in the western United States. Hutchinson Ross, Stroudsburg, Pennsylvania. (viii)

Swanson, F.J., S.L. Johnson, S.V. Gregory, and S.A. Acker. 1998. Flood disturbance in a forested mountain landscape. BioScience 48:681-689. (i)

Swanston, D.N. 1991. Natural Processes. Pages 139-179 *in* W. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society, Bethesda, Maryland. (i)

Sweeney, B.W., T.L. Bott, J.K. Jackson, L.A. Kaplan, J.D. Newbold, L.J. Standley, W.C. Hession, and R.J. Horwitz. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proceedings of the National Academy of Sciences101:14132-14137. (i)

Tabacchi, E., L. Lambs, H. Guilloy, A.M. Planty-Tabacchi, E. Muller, and H. Decamps. 2000. Impacts of riparian vegetation on hydrological processes. Hydrological Processes 14:2959-2976. (i)

Tal, M., K. Gran, A.B. Murray, C. Paola, and D.M Hicks. 2004. Riparian vegetation as a primary control on channel characteristics in multi-thread rivers. Pages 43-58 *in* A.J. Bennet and A. Simon, editors. Riparian vegetation and fluvial geomorphology. American Geophysical Union, Washington, D.C. (i)

Tal, M., and C. Paola. 2007. Dynamic single-thread channels maintained by the interaction of flow and vegetation. Geology 35:347-350. (i)

Thorn, C.E., and M.R. Welford. 1994. The equilibrium concept in geomorphology. Annals of the Association of American Geographers 84:666-696. (i)

Tooth, S., H. Rodnight, G.A. Duller, T.S. McCarthy, P.M. Marren, and D. Brandt. 2007. Chronology and controls of avulsion along a mixed bedrock-alluvial river. Geological Society of America Bulletin 119:452-461. (i)

Torgersen, C., E., C.V. Baxter, and H.W. Li. 2006. Landscape influences on longitudinal patterns of river fishes: spatially continuous analysis of fish-habitat relationships. American Fisheries Society Symposium 48:473-492. (i)

Trimble, S.W., and A.C. Mendel. 1995. The cow as a geomorphic agent—a critical review. Geomorphology 13:233-253. (i)

Van Horne, B. 1983. Density as a misleading indicator of habitat quality. Journal of Wildlife Management 47:893-901. (i)

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137. (i)

Ward J.V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83:269-278. (i)

Wells, R.R., C.V. Alonso, and S.J. Bennett. 2009. Morphodynamics of headcut development and soil erosion in upland concentrated flows. Soil Science Society of America Journal 73:521-530. (i)

Wemple, B.C., J.A. Jones, and G.E. Grant. 1996. Channel network extension by logging roads in two basins, western Cascade, Oregon. Journal of the American Water Resources Association 32:1195-1207. (i)

Whitaker, A., Y. Alila, J. Beckers, and D. Toews. 2002. Evaluating peak flow sensitivity to clear-cutting in different elevation bands of a snowmelt-dominated mountainous catchment. Water Resources Research 38:1172. (i)

School of Bull Trout Salvelinus confluentus/*William Meyer, WDFW*

Williams, G.P., and M.G. Wolman. 1985. Effects of dams and reservoirs on surface-water hydrology – changes in rivers downstream from dams. U.S. Geological Survey, Washington, D.C. (i)

Wohl, E. 2013. Mountain rivers revisited. American Geophysical Union, Washington, D.C. (i)

Wohl, E. 2014. Rivers in the landscape: science and management. Wiley, Hoboken, New Jersey (i)

Wohl, E., D. Cooper, L. Poff, F. Rahel, D. Staley, and D. Winters. 2007. Assessment of stream ecosystem function and sensitivity in the Bighorn National Forest, Wyoming. Environmental Management 40:284-302. (i)

Wolman, M.G. 1959. Factors influencing erosion of cohesive river banks. American Journal of Science 25:204-216. (i)

Wolman, M.G. 1967. A cycle of sedimentation and erosion in urban river channels. Geografiska Annaler 49A:385-395. (i)

Wright H.A., and A.W. Bailey. 1982. Fire ecology: United States and southern Canada. Wiley, New York. (i)

Ziemer, R.R., and T.E. Lisle. 1998. Hydrology. Pages 43-68 *in* R.J. Naiman and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Chapter 3. Wood

By: *George F. Wilhere and Anne Marshall*

3.1. Introduction

The scientific study of wood in rivers and streams began about 50 years ago (Gregory 2003) and the vital ecological role of instream wood has been known for at least 40 years (Swanson et al. 1976). Over that 50 year period, the substantial quantity of research on instream wood has motivated numerous authoritative reviews of the scientific literature, such as Bisson et al. (1987), Maser et al. (1988), Gurnell et al. (1995), Bilby and Bisson (1998), Gurnell et al. (2002), Naiman et al. (2002a), and Hassan et al. (2005). A comprehensive, in-depth review of the ecology of wood in aquatic ecosystems is provided by Gregory et al. (2003). Although we have a solid understanding of wood's role in aquatic ecosystems and the pathways through which wood enters and moves through a stream network, some important questions remained unanswered.

Wood, rocks, and roots – roughness elements that enhance habitat diversity by increasing hydraulic complexity/Jane Atha, WDFW

This chapter focuses on the roles and recruitment of wood in aquatic ecosystems. It does not cover stream restoration using large wood. For information on that, see Dominguez and Cederholm (2000), Reich et al. (2003), Bisson et al. (2003), Carah et al. (2014), and Roni et al. (2014). For information on riparian forest management see Boyer et al. (2003) and Spies et al. (2013).

3.2. The Ecological Role of Instream Wood

Wood plays critical roles in the composition, structure, and function of riparian and aquatic ecosystems. In forested regions, wood is an important determinant of channel form and dynamics, especially in small streams (Montgomery and Buffington 1997; Bilby and Bisson 1998). However, the importance of wood's effects differs with channel dimensions, slope, sediment supply, and stream discharge (Gurnell et al. 1995). Large wood causes widening and narrowing, deepening and shallowing, stabilization and destabilization at different points along a stream or river channel (Swanson et al. 1976). The many effects of large wood create a variety of channel morphologies—dam pools, plunge pools, riffles, glides, undercut banks, and side channels—which provide a diversity of aquatic habitats. Pools are deposition sites for sediment and fine organic matter. Sediment is essential substrate for salmonid spawning, and stored sediments become mobilized over time to replenish downstream spawning areas.

Sediment and fine organic matter deposits are productive areas for invertebrates and are important food production sites for juvenile salmonids (Bisson et al. 1987). Pools provide rearing habitats and essential refuge from high flows for fish and other aquatic fauna. Instream large wood provides fish with cover from predators, and by increasing a water body's effective space, wood structures may increase fish densities (Bisson et al. 1987).

3.2.1. Small Wood

Wood is often divided into large and small size categories. Large wood is usually defined as having diameter greater than 4 in (10 cm) and length greater than 6 ft (≈2 m) (Bilby and Ward 1991; Schuett-Hames et al. 1999).[1] Small wood[2] consists of branches and other woody material not classified as large wood. Small wood plays essential and unique roles in lotic ecosystems. For instance, accumulations of small wood enhance a stream's retention of leaves and particulate organic matter (Gregory et al. 1991), which are vital food sources for many aquatic invertebrates. Small wood also effects channel morphology. Plunge and dammed pools are often associated with accumulations of small wood, and large wood with dense accumulations of small wood retain sediment significantly more frequently than large wood with sparse small wood accumulations (Bilby and Ward 1991).

Small wood exerts tremendous effects on hydrology and channel morphology through actions of the American Beaver *Castor canadensis* (Naiman et al. 1988). Beaver dams can store large quantities of sediment, reduce channel incision, remove excess nutrients from water, increase water retention and base flows, reduce peak flows, and by spreading flow events over longer periods, increase groundwater recharge, and increase the salmonid habitat capacity of small streams (Pollock et al. 2015). Beaver dams consist mainly of mud and small wood. Published information on small wood sizes in dams of Beaver is lacking. However, Beaver most often forage on trees ranging from 1.2 - 3.2 in (3 - 8 cm) in diameter (Collen and Gibson 2001), and a large proportion of woody stems used as food are also used in dam construction (Barnes and Mallik 1996). In Maryland, Blersch and Kangas (2014) found that 98% of sticks (i.e., small wood) in a Beaver dam were less than 4 in (10 cm) in diameter and that 46% of those sticks were probably placed in the dam by Beaver. The other 54% of sticks in the dam were due to passive capture of transported wood. For more information on the role of Beaver and Beaver dams in aquatic ecosystems, see Pollock et al. (2015).

3.2.2. Roles of Instream Large Wood

The main role of large wood in aquatic ecosystems is large roughness elements (Bisson et al. 1987) (Figure 3.1). Roughness elements are obstacles in a channel that deflect flow and change its velocity. The size, shape, and strength of large wood make it very effective at redirecting hydraulic forces and the flow of materials (Figure 3.2), such as sediment and fine organic matter. Instream large wood increases hydraulic complexity, i.e., creates a wider range of flow velocities, which causes pool formation, streambed scour, sediment deposition, and channel migration. The net result is a diversity of aquatic habitats.

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[1] Large wood is also known as large woody debris (LWD) or coarse woody debris. There is no universal definition of LWD. Another definition is greater than 4 in (10 cm) in diameter and greater than 3 ft (≈1 m) in length. The system of measurement used effects the size of large wood. The definition using English units (4 in wide and 6 ft long) results in a piece of smaller volume than the definition using metric units (10 cm wide and 2 m long).

[2] Small wood is also known as small woody debris or fine woody debris

The influences of instream large wood on aquatic ecosystems are a function of wood size relative to channel size. For instance, as much as 80% of pools in small streams can be associated with wood (Montgomery et al. 1995); however, the frequency of wood-associated pools decreases with increasing stream size (Bilby and Ward 1989, 1991; Montgomery et al. 1995). The latter relationship is due to the increased capacity of larger streams to transport large wood downstream.

In low order streams, "key pieces" of large wood affect channel morphology. Key pieces are defined as large wood that is independently stable within the bankfull channel (i.e., not held or trapped by other material) and have the potential to retain other pieces of large wood (WFPB 2011). Key piece size increases as channel width increases (Table 3.1). In large rivers, wood influences channel morphology through tangled accumulations of wood known as woody debris jams.

Stable structural features can promote channel stability (Sullivan et al. 1987). In large rivers, for instance, woody debris jams contribute to floodplain stabilization by initiating the formation of mid-channel bars that eventually become forested islands (Fetherston et al. 1995; Abbe and Montgomery 1996). In small streams, persistent large wood structures trap sizable amounts of sediment thereby increasing channel stability (Figure 3.3). Of all structures capable of storing sediment (i.e., wood, boulders) in small non-fish-bearing streams of northwest Washington, 93% were composed of large wood (Grizzel and Wolff 1998). Thirty to 80% of a stream's drop in elevation can be influenced by large wood (Keller and Swanson 1979), and log steps can reduce average channel gradients by 8 to 22% (Heede 1972). Reducing channel gradient dissipates stream power which mobilizes less sediment and increases sediment storage which can subsequently increase channel stability.

Table 3.1. Minimum volume of large wood key pieces for different channel widths. Example dimensions for key piece length and diameter yield minimum volume. Diameter is measured at piece midpoint. One meter is approximately 3.3 ft.

		Example dimensions (m)	
Bankfull width (m)	Minimum volume (m^3)	Length	Diameter
0-5[a]	1	2.5	0.71
5-10[a]	2.5	7.5	0.65
10-15[a]	6	12.5	0.78
15-20[a]	9	17.5	0.81
20-30[b]	9.75	17.5	0.84
30-50[bc]	10.5	17.5	0.87
50-100[bc]	10.75	17.5	0.88

[a] WFPB (2011), [b] Fox and Bolton (2007), [c] Must have an attached root wad.

Studies where wood has been experimentally removed from streams further reveal the significant role of large wood for storing sediment. For instance, Bilby (1981) measured a 72% reduction in sediment storage following removal of large wood from 575 ft (175 m) of streambed. Beschta (1979) observed a similar effect after woody debris dams were removed from a moderate gradient headwater stream in western Oregon where 6,500 yd^3 (5,000 m^3) of sediment was lost from 820 ft (250 m) of streambed over a period of about 10 months.

Large wood can also cause transient channel instability. Keller and Swanson (1979) and Nakamura and Swanson (1993) describe stream reaches where large wood caused local erosion that increased channel width by more than 50%. In one case a woody debris jam increased channel width by 230%. These two studies also describe reaches where large wood caused lateral migration of the main channel and formation of side

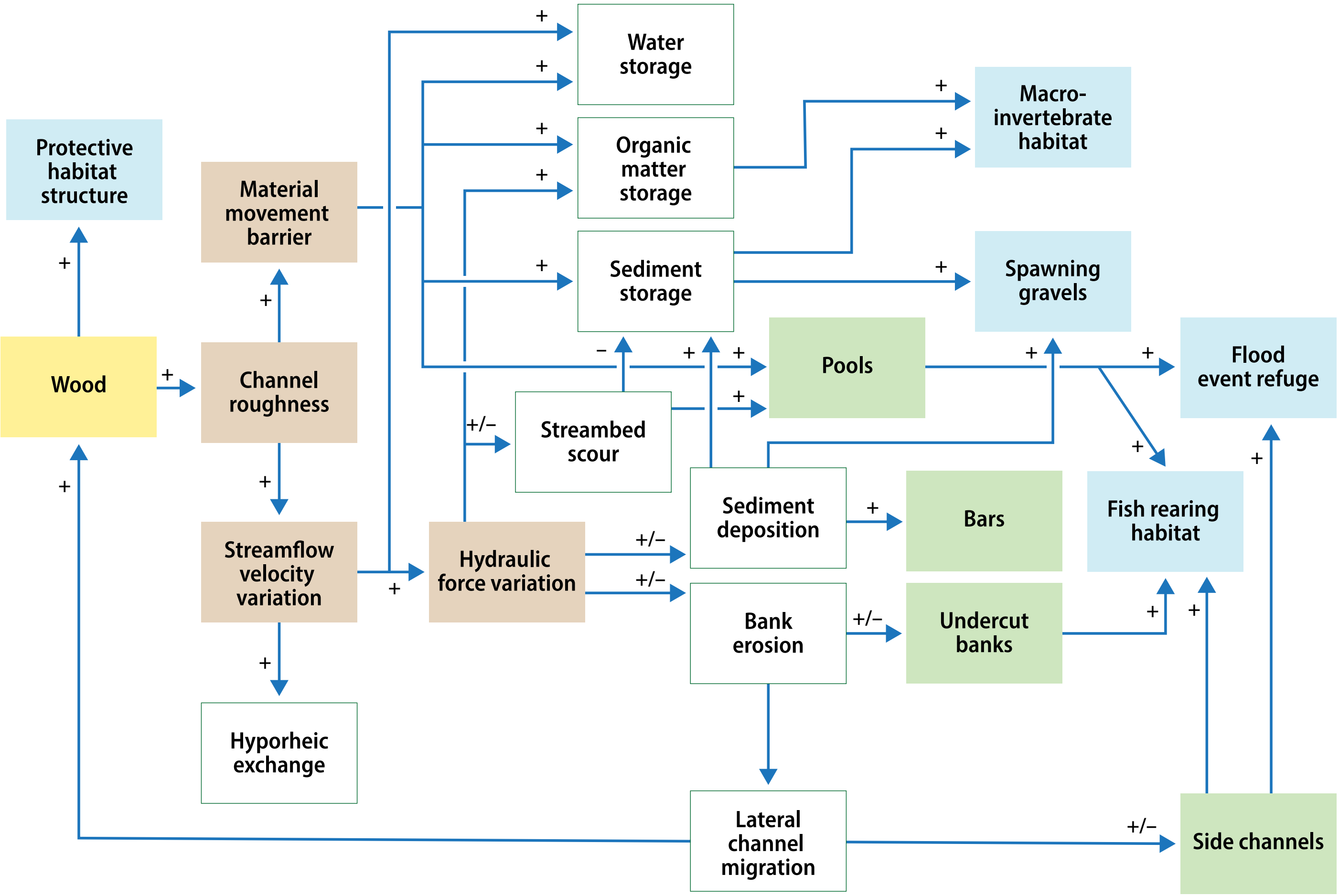


Figure 3.1. Conceptual model depicting the effects of wood on fluvial systems and fish habitats. Arrows indicate the hypothesized directions of causal relationships; pluses and minuses indicate the hypothesized slope of each relationship. Brown boxes represent direct effects of wood on fluvial processes. White boxes are secondary processes. Green boxes depict channel structures. Blue boxes are habitat features. Diagram does not include most feedback loops.

channels. Over time, these types of disturbances create a shifting mosaic of aquatic and riparian habitats. Wood influences nutrient dynamics in two major ways: wood contains nutrients that are released through decomposition, and large wood influences the rate and timing of organic materials transport (Bilby 2003). The latter is the more important effect. Instream large wood traps and retains organic matter, such as leaves and other plant detritus, which are essential to the aquatic food web. Trapped organic materials are food sources for aquatic microorganisms, such as bacteria and certain fungi, and invertebrates, such as insects, crustaceans, and mollusks.

Figure 3.2. Sediment and nutrient storage and gravel bar formation behind large wood/Jane Atha, WDFW

Figure 3.3. Large wood altering channel morphology by creating dam pool, step drop, and plunge pool/Jane Atha, WDFW

The relationship between instream large wood and nutrient storage is well established. Larger quantities of wood per unit area in small streams lead to higher rates of storage of fine organic matter (Bilby and Likens 1980; Bilby and Bisson 1998; Brookshire and Dwire 2003). The mass of coarse particulate organic matter in streams in the McKenzie River watershed of Oregon was positively related to amounts of instream wood (Naiman and Sedell 1979). Bilby (1981) reported dramatic increases in downstream export of fine particulates during periods of high discharge following woody debris removal.

Carcasses of adult anadromous salmon provide nutrients to aquatic ecosystems (Cederholm et al. 1989; Reimchen et al. 2003). On the Olympic Peninsula, a positive correlation was observed between the number of Coho Salmon *Oncorhynchus kisutch* carcasses retained in streams and the amount of large wood in the stream (Cederholm and Peterson 1985).

By creating dams, instream large wood regulates water flow and storage. For example, surface water volume increased by 168%, five years after restoration of instream large wood in a coastal Oregon stream (Crispin et al. 1993), and third-order streams in Indiana with woody debris dams held surface water 1.5 to 1.7 times longer than those with minimal large wood (Ehrman and Lamberti 1992, cited by Gurnell et al. 1995). Hydrological effects such as these may alter high flow events by reducing peak discharge and increasing the event's duration (Gurnell et al. 1995). Pools created by woody debris dams raise the water table in adjacent streambanks (Gurnell et al. 1995), and that affects riparian vegetation composition. For instance, at some sites, wet riparian soils may be maintained through seepage from pools, and wet soils are more suitable for Red Alder *Alnus rubra,* Red Ceder *Thuja plicata,* and Alaska Yellow Ceder *Chamaecyparis nootkatensis* than other tree species. Red Alder is known to contribute enormous amounts of beneficial nitrogen to soils and streams (Naiman et al. 2002b).

The hyporheic zone is the saturated sediment beneath stream channels and riparian areas where groundwater and instream water mix. Large wood diverts surface water flow into the hyporheic zone, and by trapping and storing sediments, large wood increases the volume of the hyporheic zone (Naiman et al. 2000). Greater hyporheic exchange can also reduce a streams thermal sensitivity (Chapter 4).

3.3. Recruitment of Instream Large Wood

The quantity of instream large wood is determined by the processes of wood recruitment, decomposition, and transport (Figure 3.4). "Recruitment" refers to the process of wood moving from the terrestrial environment to the stream channel. "Transport" refers to wood import to, movement within, and export from stream reaches. For comprehensive reviews of the dynamics of instream large wood, including decomposition and transport, see Benda et al. (2003), Bilby (2003), Gurnell (2003), and Piegay (2003).

3.3.1. Wood Recruitment Mechanisms

Large wood recruitment is often caused by bank erosion, windthrow, landslides, debris flows, snow avalanches, and tree mortality due to fire, ice storms, insects, and disease (Swanson et al. 1976; Maser et al. 1988). At any given site, more than one mechanism may contribute to recruitment. However, the dominant mechanism is determined by channel morphology, slope steepness, slope stability, forest composition and structure, and local wind patterns (Murphy and Koski 1989; McDade et al. 1990; May and Gresswell 2003; Johnston et al. 2011; Benda and Bigelow 2014). In large low-gradient rivers, for instance, wood is commonly recruited via bank erosion. Steep-sloped riparian forests along confined streams can recruit instream large wood via several mechanisms; however, landslides and windthrow are the dominant recruitment mechanisms in such settings (Nakamura and Swanson 1993; May and Gresswell 2003). Fox (2001) reported that smaller channels are likely to obtain a significant proportion of instream large wood by stem breakage and individual tree mortality due to insects or disease because smaller streams recruit less wood through lateral bank avulsion. Beaver recruit small wood to small, low-gradient streams (Pollock et al. 2003).

Instream wood originating from bank erosion increases as channel confinement decreases (Murphy and Koski 1989; Nakamura and Swanson 1993; Martin and Benda 2001; Johnston et al. 2011). In relatively unconfined third-order channels surrounded by mature and old-growth forests in British Columbia, bank erosion was the dominant route of large wood delivery, especially in wider channels (Johnston et al. 2011). Along Alaska's southeastern coast, bank erosion dominated large wood recruitment in alluvial channels, while windthrow was dominant in bedrock channels (Murphy and Koski 1989). Over a 63 year period, the Queets River (bankfull width = 420 ft [128 m]) recruited 95% of wood pieces ≥ 3 ft (1 m) in diameter through bank erosion that occurred as its channel meandered across its wide floodplain (Latterell and Naiman 2007).

The relative importance of each instream wood recruitment mechanism is highly variable (Table 3.2). Johnston et al. (2011) found tree mortality was the most common recruitment mechanism followed by (in descending order) bank erosion, windthrow, and landslides. In contrast, Murphy and Koski (1989) found that tree mortality was the third most common mechanism, and their rank ordering was bank erosion, windthrow, tree mortality, and landslides. Johnston et al. (2011) found that windthrow contributed 4% of instream large wood, but May and Gresswell (2003) found windthrow contributed 59%, on average. Benda et al. (2003) found that landslides contributed, on average, 17% of instream wood volume, but at the site-level, the contribution from landslides ranged from 0 to 66%. In the Oregon Coast Range, 65% of instream pieces of large wood and 46% of wood volume in a fourth order stream were delivered by landslide or debris flow (Reeves et al. 2003).

Large wood suspended over water creates a refuge for fish from avian predators/Jane Atha, WDFW

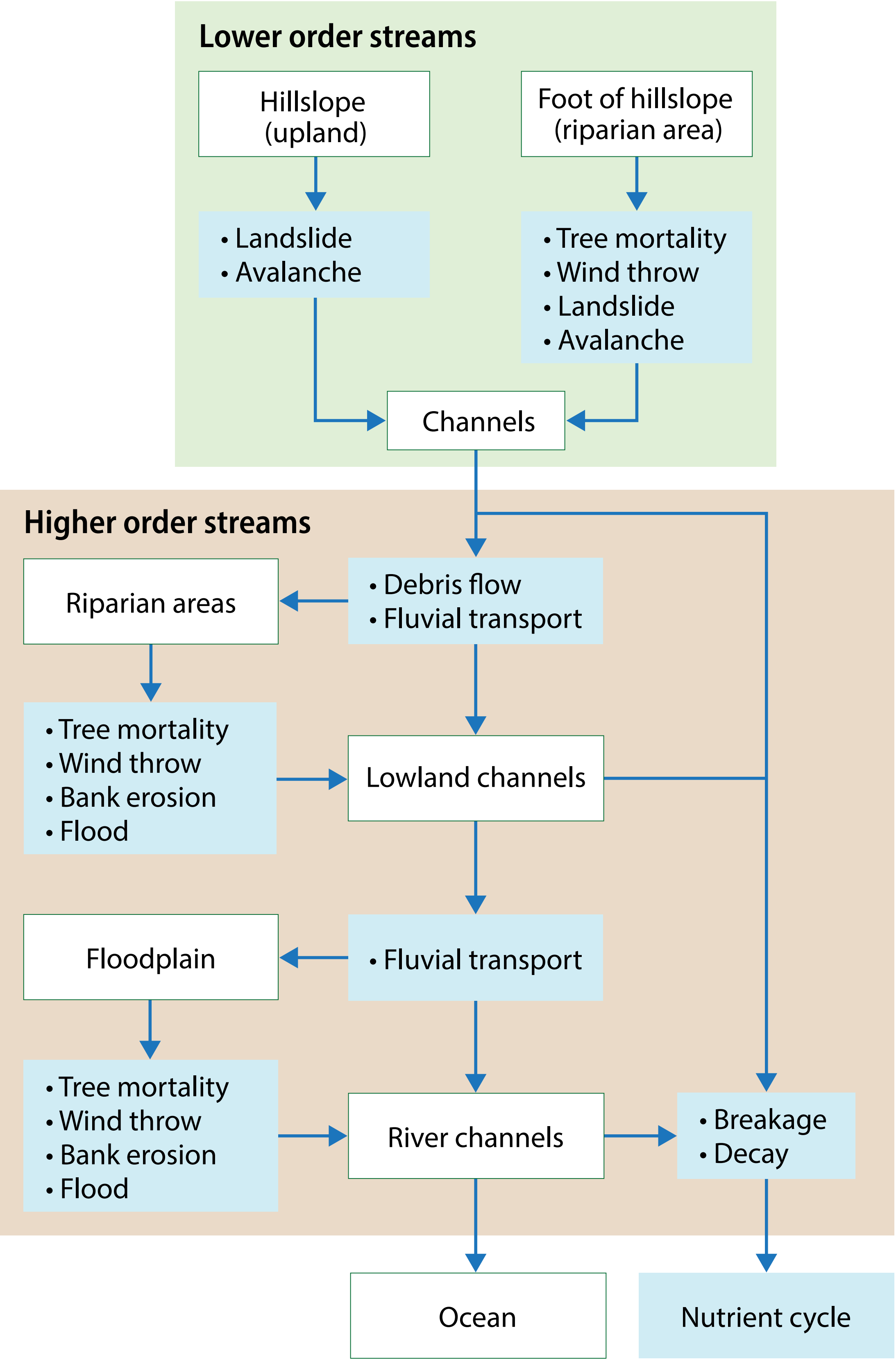


Figure 3.4. Wood recruitment mechanisms and wood transport pathways (adapted from Hassan et al. 2005). Lower order streams are first and second order headwaters. White and blue boxes are ecosystem structures and processes, respectively.

Table 3.2. Mean percentage of instream large wood contributed by various recruitment mechanisms in unmanaged old-growth conifer forests. For Johnston et al. (2011), stem breakage could be lumped with the windthrow or tree mortality categories. One meter is approximately 3.3 feet.

		Southeast Alaska[1]	South & Central British Columbia[2]	Coast Range Oregon[3]	Northern California[4]
	Stream length (m)	10,280	8,129	3,220	4,470
	Wood units	% of wood pieces	% of wood pieces	% of wood pieces	% of wood volume
Recruitment mechanism	bank erosion	42	18	6	29
	windthrow	33	4	59	--
	tree mortality	22	65	6	53
	stem break	--	12	--	--
	landslide	3	1	29	17

1 = Murphy and Koski 1989, 2 = Johnston et al. 2011, 3 = May and Gresswell 2003, 4 = Benda et al. 2003

A site's recruitment mechanisms determine the magnitude and timing of wood recruitment events. Recruitment can be continual or episodic. In large, meandering rivers wood may be recruited through continual bank erosion. Soil creep along the eroding base of a hillslope may also contribute to continual wood recruitment. However, most wood recruitment occurs through episodic events such as floods, landslides, debris flow, or windstorms. For example, if a 100-year flood causes a river to migrate across its floodplain, then a huge volume of wood that would have taken decades to be recruited under smaller flows could be recruited overnight. In smaller confined streams, the main recruitment mechanism may be windthrow, which can recruit wood incrementally (single tree per event) or catastrophically (100s of trees per event).

3.3.2. Recruitment Distances

Forests adjacent to streams provide the majority of large wood delivered to stream channels (Murphy and Koski 1989; McDade et al. 1990), however, trees far removed from riparian areas may also be delivered via landslides (Reeves et al. 2003) or channel migration (Latterell and Naiman 2007). Johnston et al. (2011) found that mean recruitment distances were greatest for wood entering by way of landslides, followed by (in descending order by distance) windthrow, stem breakage, falling of dead trees, and bank erosion.

Source distances are affected by channel geomorphology. The source distance of instream large wood differed significantly ($P < 0.05$) when comparing alluvial streams to colluvial channels draining steep hillslopes in Oregon's Coast Range (May and Gresswell 2003). In this study, 80% of wood pieces and total wood volume originated from forests within 160 ft (50 m) of colluvial channels confined by steep hillslopes, whereas in unconfined alluvial channels, 80% of instream large wood originated from within 98 ft (30 m) of the channel (May and Gresswell 2003). Along steep second growth redwood forests in northern California, landslides resulted in recruitment distances extending over 197 ft (60 m) (Benda et al. 2002). In the Oregon Coast Range, large wood from pristine steeply sloped conifer-deciduous forest was delivered to a fourth order stream by landslide or debris flow from distances of more than 295 ft (90 m) upslope of the channel (Reeves et al. 2003).

Table 3.3. Recruitment distances of instream large wood from unmanaged old-growth conifer riparian forests.

Location	Stream order or width	Maximum recruitment distance	Dominant recruitment process	Recruitment from intermediate distances	Source*
Southeast Alaska	second- to fifth-order	> 100 ft > 30 m	bank erosion	99% of pieces from < 100 ft (30 m). 45% of pieces from within 3 ft (1 m) of streambank. Average tree height = 130 ft (40 m)‡.	1
Western Oregon & Washington	first- to third-order	180 ft 55 m	not reported	50% of pieces from within 33 ft (10 m) of streambank; 85% from within 100 ft (30 m); 90% from within 128 ft (39 m), horizontal distance. Average tree height = 189 ft (57.6 m).	2
Northern California	45-55 ft 14-17 m	180 ft 55 m	variable	90% of wood from < 100 ft (30 m), slope distance. Longer recruitment distances due to landsliding.	3
Coast Range Oregon	second- and third-order	230 < d < 262 ft 70 < d < 80 m	windthrow	In alluvial stream 80% of pieces from ≤ 100 ft (30 m). In colluvial streams ≈50% of pieces from ≤ 100 ft (30 m). For both types ≈75% of volume from ≤ 165 ft (50 m).	4
Coast Range Oregon	fourth-order	> 300 ft > 90 m	landslide & debris flow	About 65% of pieces and 46% of wood volume were from upslope sources by landslides or debris flows.	5
Olympic Peninsula	sixth-order	> 1,475 ft > 450 m	channel migration	95% of wood from < 870 ft (265 m). 50% of pieces from within 300 ft (92 m).	6
Southeast Alaska	16-94 ft 5-30 m	115 < d < 130 ft 35 < d < 40 m	variable	96% of all large wood from within 65 ft (20 m) and 89% of from within 33 ft (10 m). Average tree height = 74 ft (22.6 m).	7
South & Central British Columbia	3-55 ft 1-17 m	425 ft 65 m	tree mortality	90% of large wood pieces and volume delivered to stream channels from within 33 and 30 ft (10 and 9 m), respectively. Average tree height = 120 ft (37 m).	8

* 1 = Murphy and Koski 1989, 2 = McDade et al. 1990, 3 = Benda et al. 2002; 4 = May and Gresswell 2003; 5 = Reeves et al. 2003, 6 - Latterell and Naiman 2007, 7 - Martin and Grotefendt 2007, 8 - Johnston et al. 2011.

‡ Tree height from Spence et al. (1996).

Source distances are affected by tree size. In one study, source distances for approximately 90% of instream large wood from mature and old-growth conifer riparian forest in western Washington and Oregon were within 85 and 130 ft (26 m and 40 m) of the streambank, respectively (McDade et al. 1990). The difference in source distances was largely attributed to taller trees in old-growth riparian forest (mean height = 189 ft [58 m]) compared to trees in mature riparian forest (mean height = 157 ft [48 m]). Likewise, recruitment distances measured in Alaska (e.g., Murphy and Koski 1989; Martin and Grotefendt 2007) are generally shorter than those measured in Oregon (e.g., McDade et al. 1990; May and Gresswell 2003) because trees of the same species are generally shorter in Alaska than in Oregon (Table 3.3). Johnston et al. (2011) also found that large wood source distances increased with increasing tree height. For a more thorough discussion of factors affecting source distances see Benda and Bigelow (2014).

The stereotypic recruitment function describing the amount of instream wood versus recruitment distance is nonlinear (Figure 3.5). That is, areas closer to the stream channel provide relatively more wood than areas farther from the stream channel. In southeast Alaska, for instance, 100% of large wood was recruited from within 98 ft (30 m) of the streambank, but 45% of large wood originated from within 3.3 ft (1 m) (Murphy and Koski 1989). Likewise, in a study from Oregon and Washington, 100% of large wood with an identified source was recruited from within 180 ft (55 m), but 50% of large wood with an identified source originated from 33 ft (10 m; McDade et al. 1990) (Table 3.3). The shape of the wood recruitment function, and in particular, the shape under different watershed and site-level conditions, is an important question for riparian forest management.

Although most wood is recruited from areas close to a stream (Table 3.2), areas farther from a stream cannot be discounted, especially along larger alluvial streams. Along a fourth order stream in Oregon, the sources for 65% of large wood originated on hillslopes prone to landslides and lay beyond 295 ft (90 m) from the stream channel (Reeves et al. 2003). In Washington, 50% of large wood in a fifth-order river was recruited by lateral channel migration from forests on floodplains and fluvial terraces that lay beyond 300 ft (92 m) from the riverbank (Latterell and Naiman 2007).

3.3.3. Wood Recruitment in Intensively Managed Forests

An important question for riparian area management is how does wood recruitment in second-growth or intensively managed forests differ from recruitment in unmanaged, "natural" riparian forests? The answer to this question is essential for understanding the effects of contemporary forest management on fish habitats, however, useful answers are difficult to obtain because forest practices today are much different from those of the past.

The current conditions of instream wood and riparian areas in second-growth forests are the result of past management, and over the past 100 years management practices have changed dramatically. In the early 20th century splash dams were built on small streams to sluice logs downstream (Bisson et al. 1987), a practice that flushed naturally occurring large wood from the channel. During the 1950s and 1960s wood was routinely removed from streams to "improve" fish passage (Sedell et al. 1988), and this practice continued until the 1980s (Bilby 1984). Until the 1970s, logging to the streambank was common in Oregon

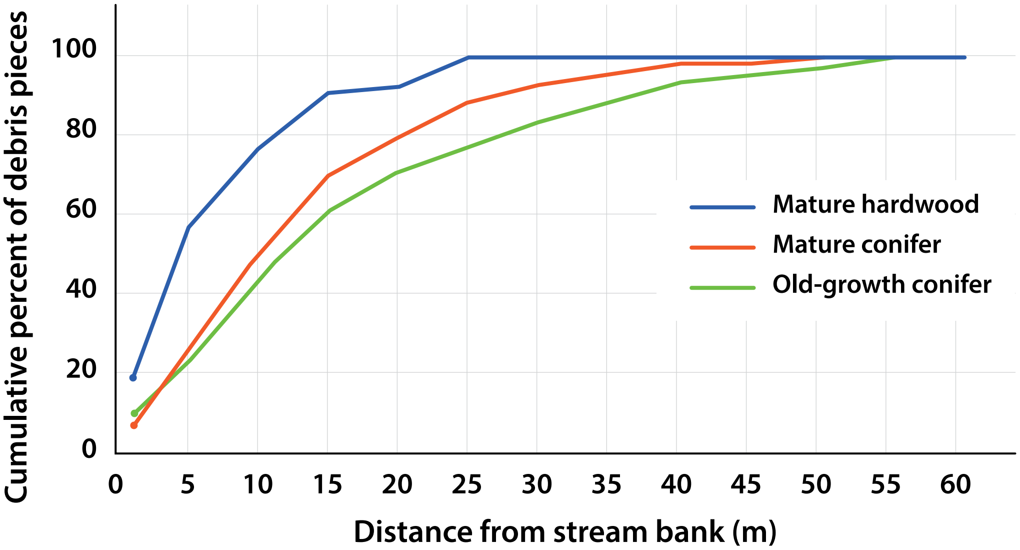


Figure 3.5. Distribution of source distances from tree origin to streambank. Empirically derived curves for old-growth conifer, mature conifer, and mature hardwood stands in western Oregon and Washington (from McDade et al. 1990). Empirically derived curves based on large wood with an identified source. One meter is equivalent to 3.3 feet.

and Washington; a practice that changed shortly after passage of the Clean Water Act of 1972. From 1987 to 2000, the Washington Forest Practices Rules required 23 to 65 trees per acre in riparian areas (WFPB 1987). In contrast, old-growth riparian areas may have 160 or more trees per acre (Acker et al. 2003). The current forest practices rule (WDNR 2005) require much wider stream-side buffers with much higher tree densities in riparian areas than pervious rules. The current regulations should result in a stand basal area equal to that of a mature conifer forest when the riparian stand is 140 years old. In short, over the past 100 years, riparian areas in intensively managed forests of Washington have been subjected to many different types of management, and over the next 100 years these same riparian areas will change dramatically.

Past forest practices are known to have severely degraded instream wood (Bilby and Ward 1991; Ralph et al. 1994), but current forest practice regulations are expected to improve instream wood (WDNR 2005). Whether current regulations will result in enough instream wood to create fish habitats that meet policy goals is unknown, and resolving that issue will be difficult until riparian areas attain their desired future conditions a century or more from now. Nevertheless, theoretical and empirical evidence suggest two main differences in large wood recruitment between second-growth, intensively managed forests and unmanaged mature and old-growth riparian forests.

Past forest practices are known to have severely degraded instream wood, but current forest practice regulations are expected to improve instream wood. Whether current regulations will result in enough instream wood to create fish habitats that meet policy goals is unknown.

First, the size of wood recruited from intensively managed forests is smaller than wood recruited from old-growth forests. Benda et al. (2002) found that the diameter of wood recruited from old-growth sites was, on average, up to twice the diameter of wood found in 50-year-old second-growth sites. Czarnomski et al. (2008) found significantly higher numbers of large wood pieces in stream segments adjacent to unmanaged mature and old-growth sites than in segments adjacent to 30- to 50-year-old intensively managed sites. From a theoretical perspective, these results are unsurprising—the dominant trees in a 200-year-old stand are much larger than the dominant trees in a 50-year-old stand. A similar relationship was also reported by McDade at al. (1990) who found that the size, both length and diameter, of instream large wood was related to forest age—wood was significantly smaller in younger forests (i.e., unmanaged mature forest versus old-growth forest).

Second, both theoretical models and empirical evidence show that maximum wood recruitment distances for intensively managed forest are less than the maximum recruitment distance for old-growth forests (McDade at al. 1990; Robison and Beschta 1990; Benda and Bigelow 2014). Again, these results are due to relative tree sizes—taller trees can contribute large wood from longer distances. The models show that the maximum recruitment distance for large wood is slightly less than the height of dominant trees; however, these models do not incorporate the recruitment processes of landslides and lateral channel migration which could substantially lengthen the maximum recruitment distance. For instance, in Benda et al. (2002) the theoretical maximum recruitment distance for their sixteen 50-year-old second-growth sites was about 98 ft (30 m), but due to landslides two sites had large wood recruited from beyond that distance—105 ft (32 m) and 210 ft (65 m). The results of Benda et al. (2002) and Benda and

Bank erosion recruits large wood into stream/Ned Pittman, WDFW

Bigelow (2014) also show that regardless of stand age or management, recruitment distances are primarily determined by recruitment processes. Riparian areas where bank erosion is the dominant process will, on average, have shorter recruitment distances, riparian areas where landslides dominate will have longer recruitment distances, and areas where other forms of tree mortality (e.g., windthrow, suppression, disease) dominate will have intermediate recruitment distances. In short, spatial variability in large wood recruitment is a function of many factors, including stream size and channel morphology, forest management, and natural disturbances such as landslides, floods, and windthrow.

3.4. Abundance of Instream Wood

3.4.1. Forested Regions of Washington

The abundance of instream large wood was much greater historically than it is today, especially in large rivers. One hundred fifty years ago, some woody debris jams on large rivers were nearly 1-mile-long (Sedell and Luchessa 1981), but these impressive natural structures were cleared for purposes of river navigation. For instance, during the latter part of the 19th century, over 5,500 pieces of wood between 5 and 9 ft (1.5 and 2.7 m) in diameter were pulled from a 50-mile (80-km) section of the Willamette River (Sedell and Luchessa 1981). Collins et al. (2002) estimated that wood in some lowland Puget Sound rivers was 10 to 100 times greater prior to European settlement.

In contrast to large rivers, the abundance of large wood in small streams of unmanaged watersheds may still be representative of historical conditions. Fox and Bolton (2007) studied the quantities of instream large wood in natural, unmanaged forested watersheds in Washington. Ninety-six percent of their 150 sites had a stand age between 200 and 800 years (Fox 2001). In these watersheds, the processes of recruitment, decomposition, and transport were presumably undisturbed by human activities. Their findings, which are presented for forest zones and channel widths, could serve as reference or target conditions for instream large wood (Table 3.4).

3.4.2. Non-forested Regions of Washington

Fox and Bolton (2007) studied forested watersheds in forested regions (sensu Franklin and Dryness 1988), and, in fact, nearly all we know about instream wood is based on studies conducted in forested watersheds. This is unfortunate because our knowledge of instream wood for non-forested regions of Washington, such as semi-arid grasslands (i.e., steppe or Palouse) and shrub-steppe, is limited. Nevertheless, instream wood was likely to have been more abundant than it is today for two reasons.

First, in basins with headwaters in forested regions, such as the Yakima, Wenatchee, and Walla Walla basins, wood was transported downstream to shrub-steppe and grassland streams. This process is known to have occurred in other semi-arid and arid regions of the United States. Minckley and Rinne (1985) present historical evidence for the movement of large wood in rivers of the desert southwest: wood originated in forested headwaters, moved sporadically through desert riparian areas during flood events, and was ultimately deposited at the mouth of the Colorado River in Mexico. Minckley and Rinne (1985) identify interception of large wood by dams as a major cause of large wood reduction in semi-arid and arid river basins. At artificial reservoirs in eastern Washington, such as Keechelus, Kachess, Cle Elum, Bumping, Clear, and Rimrock lakes, large wood that could potentially interfere with dam operations is removed and burned (W. Meyer and B. Renfrow, Washington Department of Fish and Wildlife, personal communications). Some small water diversion dams do the same with large wood. This practice diminishes the quantity and quality of fish habitat in the Yakima and Tieton rivers, especially in their semi-arid shrub-steppe sections.

Second, riparian areas in grassland and shrub-steppe regions are inhabited by a wide variety of woody plants—Black Cottonwood *Populus balsamorhiza*, Quaking Aspen *Populus tremuloides*, White Alder *Alnus rhombifolia*, Thinleaf Alder *Alnus incana*, Water Birch *Betula occidentalis*, Black Hawthorne *Crataegus douglasii*, and Yellow Willow *Salix lutea* (Crawford 2003), and the historical abundance of woody plants in riparian areas is thought to have been much greater than it is today (Wissmar et al. 1994; Kauffman et al. 1997; Wissmar 2004). Reductions in woody plant abundance are mainly due to hydrological changes caused by water diversions for irrigation (Jamieson and Braatne 2001) and livestock grazing. Restoration projects can indicate the density of woody plants that historically existed in riparian areas prior to intensive livestock grazing. Only two years after the cessation of grazing within riparian areas in northeastern Oregon, the mean crown volume of willows and Thinleaf Alder tripled in size and that of Black Cottonwood increased 9-fold (Case and Kauffman

1997). Furthermore, shrub density increased by 50%. One possible reason for the severe impacts of grazing is that riparian vegetation in the semi-arid grassland and shrub-steppe regions of Washington evolved with little grazing by large herbivores (Mack and Thompson 1982).

With the exception of Black Cottonwood, woody plants in grassland and shrub-steppe regions do not contribute key pieces of large wood to stream channels (Table 3.1).

Historically, much wood in stream channels of semi-arid regions was likely to have been small wood, and much of that small wood may have been recruited to and stored in the channel through Beaver activity. Prior to 1864, Beaver are known to have been abundant in the shrub-steppe region of central Oregon, and the eradication of Beaver from central Oregon is thought to have caused a cascade of effects: gradual disintegration of Beaver dams led to incision of streambeds, lowering of water

Table 3.4. Distributions of large wood per 100 m of stream channel by forest regions in Washington State and by bankfull width (BFW) class (Fox and Bolton 2007). Large wood was defined as pieces exceeding 10 cm (4 in) in diameter and 2 m (≈6 ft) in length. Key piece sizes are defined in Table 3.1. One meter is approximately 3.3 feet.

Forest zones	BFW Class (m)	75th Percentile	Median	25th Percentile
		Number of pieces per 100 m		
Sitka Spruce *Picea sitchensis,* Western Hemlock *Tsuga heterophylla,* Silver Fir *Abies alba,* and Mountain Hemlock *T. mertensiana*	0-6	> 38	29	< 26
	> 6-30	> 63	52	< 29
	> 30-100	> 208	106	< 57
Subalpine Fir *Abies lasiocarpa* and Grand Fir *A. grandis*	0-3	> 28	22	< 15
	> 3-30	> 56	35	< 25
	> 30-50	> 63	34	< 22
Douglas-fir *Pseudotsuga menziesii* and Ponderosa Pine *Pinus ponderosa*	0-6	> 29	15	< 5
	> 6-30	> 35	17	< 5
		Number of key pieces per 100 m		
Sitka Spruce, Western Hemlock, Silver Fir, and Mountain Hemlock	0-30	> 11	6	< 4
	> 30-100	> 4	1.3	< 1
Subalpine Fir and Grand Fir	0-15	> 4	2	< 0.5
	>15-50	> 1	0.3	< 0.5
Douglas-fir and Ponderosa Pine	0-30	> 2	0.4	< 0.5
		Volume (m^3) per 100 m		
Sitka Spruce, Western Hemlock, Silver Fir, and Mountain Hemlock	0-30	> 99	51	< 28
	> 30-100	> 317	93	< 44
Subalpine Fir and Grand Fir	0-3	> 10	8	< 3
	> 3-50	> 30	18	< 11
Douglas-fir and Ponderosa Pine	0-30	> 15	7	< 2

Wood jams are habitat features and are less common now than they were historically/Ned Pittman, WDFW

tables, loss of riparian vegetation, and subsequently deeper incision of channels (Buckley 1993). According to Pollock et al. (2007), the exact mechanism that caused widespread incision of streambeds remains uncertain, however, incision almost invariably coincided with widespread trapping of Beaver and the onset of intensive livestock grazing.

For the semi-arid grassland and shrub-steppe regions of Washington, we currently lack data with which to describe reference conditions for instream wood and for riparian plant communities. Historical reconstruction using General Land Office survey notes and historical photos is one way to develop qualitative descriptions of riparian plant communities (McAllister 2008); however, more quantitative descriptions would benefit development of management objectives. Much remains unknown about historical conditions of riparian areas in non-forested regions.

3.5. Fish, Wildlife, and Instream Wood

Authoritative reviews of the scientific literature regarding relationships between instream wood and fish or wildlife species have been written by Bisson et al. (1987), Benke and Wallace (2003), Dolloff and Warren (2003), Zalewski et al. (2003), Steel et al. (2003), and Wondzell and Bisson (2003). Certain small mammal and bird species are known to be associated with wood in stream channels (Steel et al. 2003). However, we discuss only fish, amphibians, reptiles, and invertebrates; taxonomic groups known to have strong associations with instream wood.

3.5.1. Fish

For salmonids, especially juvenile Coho Salmon, there is no more important structural component than instream wood (Bisson et al. 1987). In Puget Sound lowland streams, measures of salmonid rearing habitat were strongly linked to instream large wood abundance (May et al. 1997). In headwater streams, step pools formed by large wood were important for Dolly Varden *Salvelinus malma,* juvenile Coho Salmon, steelhead, and Cutthroat Trout *Oncorhynchus clarkii* in reaches with gradients less than 4% to greater than 10% (Bryant et al. 2007). Adult spawning Sockeye Salmon *Oncorhynchus nerka* densities were positively correlated with cover provided by large wood and undercut banks, pool area, and large wood (Braun and Reynolds 2011). The instream abundance of juvenile salmonids often is directly related to the amount of large wood (Murphy et al. 1986; Bisson et al. 1987). Stream reaches where large wood was artificially added were used by higher densities of juvenile Coho Salmon in summer and winter, and Cutthroat Trout and steelhead in winter in 30 western Oregon and Washington streams when compared to reaches where large wood was not enhanced (Roni and Quinn 2001). Spawning Coho Salmon abundance increased after instream wood structures were restored (Crispin et al. 1993). A study in British Columbia found that the biomass of yearling and older salmonids was positively correlated with stream pool volume ($R^2 = 0.92$) and that over 70% of pool volume was formed by large wood (Fausch and Northcote 1992).

Large wood provides both direct and indirect benefits to fish (Bisson et al. 1987). Indirect benefits are related to wood's role as a roughness element, and this role is especially obvious in smaller streams where wood can bridge much of a channel (Zalewski et al. 2003). Large wood within a small channel dissipates flow velocity, which reduces the energy fish expend as they move upstream (Zalewski et al. 2003). Bisson et al. (1987) state that sediment storage by large wood benefits fish primarily by buffering the stream network against rapid changes in sediment load that would degrade habitats, such as rapid sediment increases due to landslides. Wood also benefits fish by increasing habitat diversity. In seeking a path around large wood obstructions, water creates complex hydraulic patterns that carve pools and side channels, form falls, and enhance channel sinuosity, imposing numerous variations in a stream's hydrology and geomorphology. Another indirect benefit of instream large wood is its influences on the availability of prey for fish in the form of macroinvertebrates. Large wood provides habitat for invertebrates, which then supplies fish with a source of food (Bisson et al. 1987).

The direct benefits of instream large wood to fish are also important. Large pieces of instream wood provide fish with refuge from high velocity flows during floods and provide hiding cover from predators (Sedell et al. 1985; Bisson et al. 1987; Braun and Reynolds 2011). Upturned tree roots and logs were the most common type of cover used by juvenile Coho Salmon and steelhead in an unlogged, west coast Vancouver Island stream (Bustard and Narver 1975). The deep pools often formed around these features made for good cover, particularly for older juveniles (Bustard and Narver 1975). After logging, almost all Coho Salmon that remained in channels over the winter sheltered in stream segments jammed with logs, with undercut banks, and in pools filled with upturned tree roots and other forest debris (Tschaplinski and Harman 1983). Hartman (1965) also found overwintering Coho Salmon fry preferring cover near logs, roots, and banks in streams in coastal British Columbia. Large wood increases fish density by visually isolating individual fish, which reduces inter- and intra-species competition (Dolloff and Reeves 1990; Crook and Robertson 1999).

3.5.2. Amphibians and Reptiles

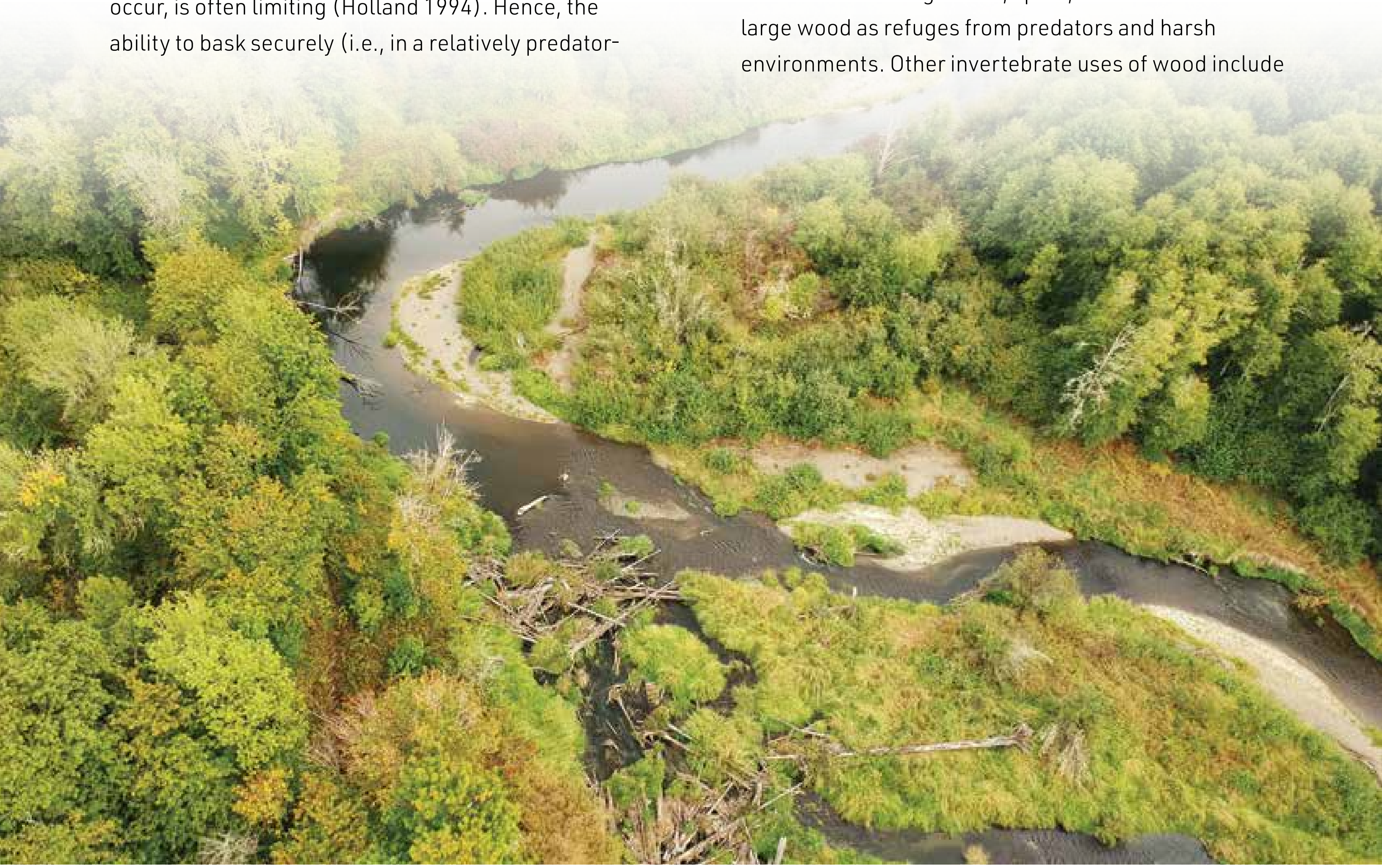
Amphibians can be a dominant biotic component in Pacific Northwest streams (Olson et al. 2007), but in contrast to fishes and invertebrates, studies that characterize the importance of instream large wood for amphibians are sparse (Wondzell and Bisson 2003). However, diverse sources of data on stream-associated amphibian species life histories indicate that instream wood contributes to the creation and maintenance of breeding, rearing, and foraging habitats, and likely overwintering habitats as well. In particular, large wood can serve as the instream substrate for amphibian oviposition (Henry and Twitty 1940; Jones et al. 1990). Wood also forms steps in streams that promote sediment wedges (May and Gresswell 2003) that serve as variable but often extensive habitat matrices for oviposition and rearing (Nussbaum 1969; Wilkins and Peterson 2000). Moreover, wood-formed steps often create dam and plunge pools that are important for amphibian foraging or refuge (Wilkins and Peterson 2000; Welsh and Lind 2002). The basis of the former is that many aquatic invertebrates eaten by stream-associated amphibians occur in the habitat matrices created by wood-formed steps (Parker 1994; Steele and Brammer 2006; see next section). In coastal Washington headwater streams, the density of stream-associated amphibians

Van Dyke's Salamander Plethodon vandykei *use large wood for cover/ Eric Lund, WDFW* was positively correlated with the amount of functional large wood ($R^2 = 0.50$), and torrent salamander (family Rhyacotritonidae) densities were positively correlated with the percentage of pools, indicating the importance of habitat complexity that large wood creates (Jackson et al. 2007). In Olympic National Park, large wood was a significant variable in explaining the densities of all three stream-associated amphibian species that were present, but its relative importance varied between species and was less important than one or more of aspect, gradient, and substrate in habitat use models for tailed frogs (family Ascaphidae) and torrent salamanders (Adams and Bury 2002).

Instream large wood is especially important for one Pacific Northwest reptile order: turtles. In the Northwest, water temperatures, even in the lowland stillwater habitats in which our two native turtle species occur, is often limiting (Holland 1994). Hence, the ability to bask securely (i.e., in a relatively predator-free environment) at temperatures that exceed those of water is critical to the normal processes of digestion and for female turtles, successfully yolking up eggs. Large floating or anchored-downed logs with enough area extending above the water surface are the key structures on which basking can occur (Holland 1994). Such structures are especially important in stillwater side- and off-channel habitats of large streams.

3.5.3. Aquatic Invertebrates

The distribution and abundance of aquatic invertebrates are influenced by instream wood and certain taxa are dependent on instream wood. In streams of all sizes, areas of wood accumulation are often also invertebrate biodiversity hot spots (Benke and Wallace 2003; Wondzell and Bisson 2003). Many aquatic invertebrates use the numerous grooves, splits, and fissures in large wood as refuges from predators and harsh environments. Other invertebrate uses of wood include



Large wood in Deschutes River/Noll Steinweg, WDFW oviposition, case making, and emergence, especially when submerged pieces are decayed rather than firm (Harmon et al. 1986). Several families of caddisfly use wood for case construction. Of ninety-two case-making caddisfly genera reviewed by Wiggins (1977), roughly a quarter exploit bark or wood to varying degrees for case construction. Instream wood also is used as sites for oviposition above and below the water line. Many caddisflies in the family Limnephilidae deposit their egg masses on moist wood. In the Coast Range of Oregon, Wisseman and Anderson (1984) found oviposition by a number of caddisfly taxa on overhanging logs. The surfaces of instream wood also are used as nursery areas for early instars, and for resting, molting, and pupation (Anderson et al. 1978).

Large wood is especially important as substrate for invertebrates to attach to in rivers with limited sources of stable substrate (Benke and Wallace 2003). Trees that have fallen into the main channels of low-gradient rivers frequently are the only stable habitat (Benke and Wallace 2003). In these streams, wood can be especially important for periphyton and macroinvertebrate taxa to attach themselves (Benke and Wallace 2003). In contrast to smaller rivers, instream large wood shifts to channel margins and floodplains of larger rivers (Bisson et al. 1987), and hence, in these main-stem locations invertebrate densities peak (Ward et al. 1982). In headwater streams, large wood and gravel habitats supported higher densities and biomass of benthic invertebrates than cobble habitats (Hernandez et al. 2005). Coe et al. (2009) saw significantly higher densities of invertebrates in Pacific Northwest streams where engineered logjams were added. They attributed this to increased overall habitat surface area, and thereby the potential for increased productivity relative to reaches with low levels of wood. Instream large wood also supported a richer and more diverse invertebrate fauna than either cobble or gravel substrates.


Large wood on Cedar River/Hal Beecher, WDFW

Wood provides invertebrates with access to a wide variety of food sources. Large wood traps and stores organic matter, and wood can be directly consumed by some invertebrates (Benke and Wallace 2003). Trapped organic materials are food sources for bacteria, fungi, mollusks, insects, and crustaceans. Although only a small fraction of aquatic insect taxa exploit woody debris as a source of food (Harmon et al. 1986), direct wood consumption by aquatic invertebrates in the Pacific Northwest has been observed in some species of caddisfly, mayfly, beetles, and snails (Anderson et al. 1978; Dudley and Anderson 1982; Pereira and Anderson 1982; Pereira et al. 1982). Submerged large wood may also support surface films of micro-organisms (e.g., diatoms, bacteria, fungi) that efficiently take up instream nitrogen and form a food web base for aquatic invertebrates (Dudley and Anderson 1982; Johnson et al. 2003), that in turn support vertebrate species (Ashkenas et al. 2004). By moderating pulses of fine sediment (Entrekin et al. 2009), instream large wood also acts to reduce the accumulation of sediment on algae (Richardson 2008), which is an important food to many macroinvertebrate taxa.

3.5.4. Fish and Wildlife Summary

The vital ecological role of instream wood has been known for at least 40 years (Swanson et al. 1976). Relationships between high quality salmonid habitats and instream large wood are well established (Bisson et al. 1987; Dollof and Warren 2003). Invertebrate taxa, such as aquatic insects, are also known to rely on large wood as a habitat substrate and food source. Juvenile salmonids prey on aquatic insects. Amphibians and turtles also depend on the channel morphology and wood structure created by instream large wood.

3.6. Land Use Effects

The major human land uses in Washington—urban/suburban, agriculture, and forestry—have altered or removed extensive areas of riparian forest and have adversely impacted instream wood size, abundance, and distribution, and left many streams with a chronic deficiency of instream large wood compared to historical conditions (Maser et al. 1988; Bilby and Ward 1991; Stouder et al. 1997; May et al. 1997).

In general, the three major land uses impact instream large wood in similar ways—they diminish the amount of instream wood by reducing the number and size of trees in riparian areas. Furthermore, large wood is often purposefully removed from rivers and streams because it can block culverts or damage other human-made structures such as bridges, dams, or levees; cause streambank erosion that destroys private property; and present hazards to navigation or recreation. Culverts and dams also disrupt the transport of wood through a stream network and reduce the amount of wood in higher order streams and rivers.

Juvenile fish use large wood for shelter/WDFW

Most research on instream wood and riparian forests has occurred in unmanaged and managed forests, but much of the knowledge gained in forests—such as the functions of instream wood, natural amounts of instream wood, and recruitment mechanisms—is applicable to riparian areas in urban/suburban and agricultural settings as well, especially in forested ecoregions.

3.6.1. Forestry

Because mature conifer trees in the Pacific Northwest have large-diameter, decay-resistant boles, streams in this region have naturally high volumes of large wood compared to other parts of North America (Harmon et al. 1986). Nevertheless, harvest in and around riparian areas can result in immediate and long-term changes to the volume of instream wood. Czarnomski et al. (2008) found that instream wood volume and abundance varied significantly relative to timber harvest and adjacent tree stand age. They found instream wood volume and abundance were significantly higher in streams adjacent to unmanaged, mature and old-growth stands compared to streams along 30- to 50-year-old plantations, and significantly less in streams adjacent to 30- to 50-year-old plantations compared to those adjacent to 20- to 30-year-old plantations or mature and old-growth forest. Beechie et al. (2000) estimated that if a riparian buffer was established today in a 50-year-old stand of Douglas-fir, recovery of pool-forming large wood might take less than 100 years in small channels (e.g., ≤ 66 ft [20 m] wide), but could take as much as 200 years in larger channels.

The impacts of logging on instream large wood and aquatic habitats may not be realized for many decades after logging. In interior British Columbia, Chen et al. (2005) found that in the short term, instream large wood volume and biomass increased after logging. However, they concluded that in the long term, abundance of instream large wood may be greatly reduced as a result of increased rates of decay, transport, and reduced recruitment from the adjacent riparian forest. With instream wood residence times averaging about 30 years, Hyatt and Naiman (2001) estimated that harvesting large riparian conifers would adversely impact aquatic habitats of large streams in about three to five decades. Murphy and Koski (1989) showed a 70% reduction of instream large wood 90 years after clearcutting without a riparian buffer. They also estimated a 250-year recovery time to pre-harvest levels.


Harvest near riparian areas can impact aquatic habitats/Marc Hayes, WDFW

The loss of riparian trees can come with long-term repercussions to functional aquatic habitat. Where riparian trees are absent, Beechie et al. (2000) predicted a lag of 7-49 and 15-91 years before pool-forming large wood could be recruited from Red Alder and Douglas-fir forests, respectively. The lag time for a given stream depends on multiple factors including channel width and the minimum size of pool-forming wood. In Alberta, instream large wood generated from stand-replacing fires was depleted within 100 years of the event but once the forests reestablished, new large wood recruitment was delayed by roughly 40 years (Powell et al. 2009).

A riparian management zone (RMZ) adjacent to logging differs in its ability to supply instream wood compared to riparian areas adjacent to unlogged uplands. On average, tree mortality was 50% greater in RMZs compared to riparian areas in unlogged landscapes, 3 to 15 years post-harvest (Martin and Grotefendt 2007). Differences in stand mortality between logged and unlogged tracts were primarily due to greater windthrow at a small proportion of RMZs in logged tracts. Additionally, downed tree recruitment from RMZ outer zones was more than double that of trees recruited in unlogged tracts. Consequently, wood recruitment increased by an average of 2 trees per 330 ft (100 m) in the buffered versus unlogged forested tracts (Martin and Grotefendt 2007).

Trees in RMZs with adjacent clear-cuts may be more susceptible to windthrow. Windthrow in RMZs (85 ft [26 m] wide on both side of streams) increased instream large wood by 34% within one to three years after logging in small streams (Grizzel and Wolff 1998). The authors of that study suggested that the RMZs adjacent to clearcuts served as a long-term source of large wood. They added that over a 10-year period, windthrow could add an average of 1.9 trees per 984 ft (300 m) of stream. Along second to fifth order streams, windthrow in RMZs was 26 times greater up to three years post-harvest compared to estimated mortality rates of trees in riparian stands in unlogged tracts (Liquori 2006). Post-harvest windthrow may reduce tree density in RMZs enough to significantly reduce competition-induced tree mortality, which in turn may lead to substantially different wood recruitment dynamics in buffers compared to unlogged forests (Liquori 2006).

3.6.2. Urban/Suburban

Very few studies have examined the impacts of development on instream large wood dynamics. In their analysis of the effects of urbanization, May et al. (1997) noted that as a basin's urbanization increased the quantity of large wood declined, as did related measures of habitat quality, namely loss of riparian forest area, reduced pool area, and decreased habitat complexity. Urban streams greater than 2% gradient and lacking large wood were found to be more susceptible to scour than less developed counterparts. Other than restoration sites where instream large wood had been replaced, high quantities of instream large wood were only found in undeveloped watersheds.

An analysis of relationships of channel characteristics, land ownership, land use, and land cover to instream large wood abundance in western Oregon showed that the most important predictor for large wood volume was land ownership, followed by stream gradient (Wing and Skaugset 2002). They also observed fewer large wood pieces in streams near rural residential areas compared to managed forests. In a study outside the Pacific Northwest, researchers in the Piedmont region of Georgia found that the absence of forest cover in catchments in suburban Atlanta corresponded to a decrease of instream large wood biomass (Roy et al. 2005).

Agricultural area in western Washington/Marlin Greene, One Earth Images

3.6.3. Agriculture

Many types of agriculture remove riparian forest, and consequently, reduce the recruitment of large wood to fish-bearing rivers and streams. Within the Nooksack River basin of northwest Washington, riparian forest along reaches accessible to anadromous fish were evaluated based on their ability to contribute instream wood large enough to form pools (Hyatt et al. 2004). Seventy-four percent of this basin's riparian forests that failed to meet this criterion were found in lowland rural-agricultural areas. In contrast, only 20% of the stands that failed to meet this threshold were in federal and commercial forestlands. This contrast is notable given that agricultural and rural areas comprised only about 22% of Nooksack basin (Hyatt et al. 2004). Similar results were observed in an analysis relating land use and land cover to instream wood abundance in western Oregon (Wing and Skaugset 2002). In this study, they found fewer pieces of large wood in streams near agricultural land uses compared to that of managed forestlands.

In shrub-steppe and grassland ecoregions of eastern Washington, livestock grazing in riparian areas significantly reduces the amount of woody vegetation. Significant reductions in woody vegetation lead to numerous adverse impacts to aquatic habitats: increased streambank erosion, increased overland flow and erosion within riparian areas, increased turbidity, reduced shade, higher water temperatures, loss of cover for fish due to reductions in overhanging vegetation and undercut banks, reduction in the amount of instream small wood, changes to channel morphology such as fewer pools, fewer meanders, and channel incision, lowering the water table, and reduction in essential nutrients derived from detritus (Belsky et al. 1999).

3.6.4. Land Use Effects Summary

Land uses that affect riparian forest also affect instream large wood. Of the three major land uses, contemporary forestry has the least effect on instream large wood. However, for all three land uses, the severity of effects depends largely on how riparian areas are managed.

3.7. Conclusions

Many effects of wood on aquatic ecosystems are well understood and not subtle, and therefore, ecologists are very confident about the critical role of wood in fish habitats. Ecologists are also confident about the role of riparian forests in supplying large wood to aquatic systems. Successful conservation of fish habitats and fish species in the forested regions of Washington depends on riparian forest management that delivers adequate wood to aquatic ecosystems. From a management perspective, the main scientific uncertainties are: 1) the shape of the wood recruitment function (e.g., Figure 3.5), especially the shape under different watershed and site-level conditions; and 2) the potential contributions from recruitment mechanisms outside the riparian forest, such as landslides, debris flows, or extreme channel migration. Addressing these uncertainties through research could lead to more ecologically effective and economically efficient management of riparian areas.

To maintain or restore 100% of a site's large wood recruitment potential, the site's composition and structure should be similar to that of an unmanaged riparian forest. Given current uncertainties, re-establishing riparian forest similar to unmanaged, mature or old-growth forests should be the most reliable way to provide instream large wood sizes and abundance that are similar to historical levels.

An important management question is: what are source distances for large wood recruitment from riparian areas to aquatic ecosystems? We know that source distances for wood in riparian areas are a function of tree height—source distances are longer for riparian forests with taller trees. We also know that tree heights for old-growth conifer forests can exceed 200 ft (60 m). In Washington State, Fox (2003) found that mean heights of canopy trees in old-growth riparian areas (i.e., stand age > 200 years old) ranged from 100 to 240 ft (30 to 73 m). Height variation among riparian areas was largely explained by forest zone and site productivity class. If large wood has a minimum diameter of 4 in (10 cm), then only that portion of a tree's stem greater than 4 in (10 cm) in diameter is large wood. From tree taper equations for Douglas-fir (Biging 1984), we know that for trees between 100 to 240 ft (30 and 73 m) tall about 15 to 3 percent of the stem is not large wood, respectively.[3] In other words, for the purpose of large wood recruitment, the "effective" tree height (*sensu* Robison and Beschta. 1990; Van Sickle and Gregory 1990) ranges from 85 to 230 ft (26 to 70 m), depending on site productivity class. Therefore, in forested ecoregions, 100% of a site's large wood recruitment potential, is located about 85 to 230 ft (26 to 70 m) from the stream channel, depending on site productivity class. These source distances only account for trees recruited via bank erosion, windthrow, or tree mortality. It does not account for large wood recruited to the stream channel through landslides or debris flows, nor does it account for recruitment through extreme channel migration in large river floodplains. In addition, this does not account for source distances of small wood. Source distances for small wood equal a full tree height from the stream channel.

Ecologists are confident about the critical role of wood in creating fish habitat and the role of riparian forests in supplying large wood to aquatic systems.

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[3] Effective tree height calculations were done in Microsoft Excel with equations from Robison and Beschta (1990) and Biging (1984).

3.8. Literature Cited[4]

Abbe, T.B., and D.R. Montgomery. 1996. Large woody debris jams, channel hydraulics and habitat formation in large rivers. Regulated Rivers: Research & Management 12:201-221. (i)

Acker, S.A., S.V. Gregory, G. Lienkaemper, W.A. McKee, F.J. Swanson, and S.D. Miler. 2003. Composition, complexity, and tree mortality in riparian forests in the central western Cascades of Oregon. Forest Ecology and Management 173:293-308. (i)

Adams, M. J., and R. B. Bury. 2002. The endemic headwater stream amphibians of the American Northwest: associations with environmental gradients in a large forested preserve. Global Ecology & Biogeography 11:169-178. (i)

Anderson, N.H., J.R. Sedell, L.M. Roberts, and F.J. Triska. 1978. The role of aquatic invertebrates in processing of wood debris in coniferous forest streams. The American Midland Naturalist 100:64–82. (i)

Barnes, D.M., and A.U. Mallik. 1996. Use of woody plants in construction of Beaver dams in northern Ontario. Canadian Journal of Zoology 74:1781-1786. (i)

Beechie, T.J., G. Pess; P. Kennard, R.E. Bilby, and S. Bolton. 2000. Modeling recovery rates and pathways for woody debris recruitment in northwestern Washington streams. North American Journal of Fisheries Management 20:436-452. (i)

Belsky, A.J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419-431. (i)

Benda, L., and P. Bigelow. 2014. On the patterns and processes of wood in northern California streams. Geomorphology 209:79-97. (i)

Benda, L.E., P. Bigelow, and T.M. Worsley. 2002. Recruitment of wood to streams in old-growth and second-growth redwood forests, northern California, U.S.A. Canadian Journal of Forest Research 32:1460-1477. (i)

Benda, L., D. Miller, J. Sias, D. Martin, R. Bilby, C. Veldhuisen, and T. Dunne. 2003. Wood recruitment processes and wood budgeting. Pages 49-74 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Benke, A.C., and J.B. Wallace. 2003. Influence of wood on invertebrate communities in streams and rivers. Pages 149-178 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Beschta, R.L. 1979. Debris removal and its effects on sedimentation in an Oregon Coast Range stream. Northwest Science 53:71-77. (i)

Biging, G.S. 1984. Taper equations for second-growth mixed conifers of northern California. Forest Science 30:1103-1117. (i)

Bilby, R.E. 1981. Role of organic debris dams in regulating the export of dissolved and particulate matter from a forested watershed. Ecology 62:1234-1243. (i)

Bilby, R.E. 1984. Removal of woody debris may affect stream channel stability. Journal of Forestry 82:609-613. (i)

Bilby, R.E. 2003. Decomposition and nutrient dynamics of wood in streams and rivers. Pages 135-148 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Bilby, R.E., and P.A. Bisson. 1998. Function and distribution of large woody debris. Pages 324-346 *in* R.J. Naiman and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Bilby, R.E., and G.E. Likens. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. Ecology 61:1107-1113. (i)

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[4] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Bilby, R.E., and J.W. Ward. 1989. Changes in characteristics and function of woody debris with increasing size of streams in western Washington. Transactions of the American Fisheries Society 118:368-378. (i)

Bilby, R.E., and J.W. Ward. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and second-growth forests in southwestern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48:2499-2508. (i)

Bisson, P.A., R.E. Bilby, M.D. Bryant, C.A. Dollof, G.B. Grette, R.A. House, M.L. Murphy, K.V. Koski, and J.R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. Pages 143-190 *in* E.O. Salo, and T.W. Cundy, editors. Streamside management: forestry and fishery interactions. Contribution No. 57. Institute of Forest Resources, University of Washington, Seattle. (viii)

Bisson, P.A., S.M. Wondzell, G.H. Reeves, and S.V. Gregory. 2003. Trends in using wood to restore aquatic habitats and fish communities in western North America. Pages 391-406 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Blersch, D.M., and P.C. Kangas. 2014. Signatures of self-assembly in size distributions of wood members in dam structures of *Castor canadensis.* Global Ecology and Conservation 2:204-213. (i)

Boyer, K.L., D.R. Berg and S.V. Gregory. 2003. Riparian management for wood in rivers. Pages 407-420 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Brookshire, E.N.J., and K.A. Dwire. 2003. Controls on patterns of coarse organic particle retention in headwater streams. Journal of the North American Benthological Society 22:17-34. (i)

Braun, D.C., and J.D. Reynolds. 2011. Relationships between habitat characteristics and breeding population densities in Sockeye Salmon. Canadian Journal of Fisheries and Aquatic Sciences 68:758-767. (i)

Bryant, M.D., T. Gomi, and J.J. Piccolo. 2007. Structures linking physical and biological processes in headwater streams of the Maybeso Watershed, southeast Alaska. Forest Science 53:371-383. (i)

Buckley, G.L. 1993. Desertification of the Camp Creek Drainage in central Oregon. Yearbook of the Association of Pacific Coast Geographers 55:91-126. (i)

Bustard, D.R., and D.W. Narver. 1975. Aspects of the winter ecology of juvenile Coho Salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Research Board of Canada 32:667-680. (i)

Carah, J.K., C.C. Blencowe, D.W. Wright, and L.A. Bolton. 2014. Low-cost restoration techniques for rapidly increasing wood cover in coastal Coho Salmon streams. North American Journal of Fisheries Management 34(5):1003-1013. (i)

Case, T.L., and J.B. Kauffman. 1997. Wild ungulate influence on the recovery of willows, Black Cottonwood, and Thin-leaf Alder following cessation of cattle grazing in northeastern Oregon. Northwest Science 71:115-126. (i)

Cederholm, C.J., D.B. Houston, D.L. Cole, and W.J. Scarlett. 1989. Fate of Coho Salmon (*Oncorhynchus kisutch*) carcasses in spawning streams. Canadian Journal of Fisheries and Aquatic Sciences 46:1347-1355. (i)

Cederholm, C.J., and N.P. Peterson. 1985. The retention of Coho Salmon (*Oncorhynchus kisutch*) carcasses by organic debris in small streams. Canadian Journal of Fisheries and Aquatic Sciences 42:1222-1225. (i)

Chen, X., X. Wei, and R. Scherer. 2005. Influence of wildfire and harvest on biomass, carbon pool, and decomposition of large woody debris in forested streams of southern interior British Columbia. Forest Ecology and Management 208:101-114. (i)

Coe, H.J., P.M. Kiffney, G.R. Pess, K.K. Kloehn, and M.L. McHenry. 2009. Periphyton and invertebrate response to wood placement in large Pacific coastal rivers. River Research and Applications 25:1025-1035. (i)

Collen, P., and R.J. Gibson. 2001. The general ecology of Beavers (*Castor* spp.), as related to their influence in stream ecosystems and riparian habitats, and the subsequent effects on fish—a review. Reviews in Fish Biology and Fisheries 10:439-461. (i)

Collins, B.D., D.R. Montgomery, and A.D. Haas 2002. Historical changes in the distribution and functions of large wood in Puget lowland rivers. Canadian Journal of Fisheries and Aquatic Sciences 59:66-76. (i)

Crawford, R.C. 2003. Riparian vegetation classification of the Columbia Basin, Washington. Natural Heritage Program Report 2003-03. Washington Department Natural Resources. Olympia, Washington. (viii)

Crispin, V., R. House, and D. Roberts. 1993. Changes in instream habitat, large woody debris, and salmon habitat after the restructuring of a coastal Oregon stream. North American Journal of Fisheries Management 13:96-102. (i)

Crook, D A., and A.I. Robertson. 1999. Relationship between riverine fish and woody debris: implications for lowland rivers. Marine and Freshwater Research 50:941-953. (i)

Czarnomski, N.M., D.M. David, K.U. Snyder, J.A. Jones, and F.J. Swanson. 2008. Dynamics of wood in stream networks of the western Cascades Range, Oregon. Canadian Journal of Forest Research 38:2236-2248. (i)

Dolloff, C.A., and C.H. Reeves. 1990. Microhabitat partitioning among stream-dwelling juvenile Coho Salmon, *Oncorhynchus kisutch,* and Dolly Varden, *Salvelinus malma.* Canadian Journal of Fisheries and Aquatic Sciences 47:2297-2306. (i)

Dolloff, C.A., and M.L. Warren. 2003. Fish relationships with large wood in small streams. Pages 179-194 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Dominguez, L.G., and C.J. Cederholm. 2000. Rehabilitating stream channels using large woody debris with considerations for salmonid life history and fluvial geomorphic processes. Pages 545-563 *in* E.E. Knudsen, C.R. Steward, D.D. MacDonald, J.E. Williams, and D.W. Reiser, editors. Sustainable fisheries management: Pacific salmon. Lewis Publishers, New York. (i)

Dudley T., and N.H. Anderson. 1982. A survey of invertebrates associated with wood debris in aquatic habitats. Melanderia 39:1-21. (i)

Entrekin, S.A., J.L. Tank, E.J. Rso-Marshall, T.J. Hoellein, and G.A. Lamberti. 2009. Response of secondary production by macroinvertebrates to large wood addition in three Michigan streams. Freshwater Biology 54:1741-1758. (i)

Ehrman, T.P., and G.A. Lamberti. 1992. Hydraulic and particulate matter retention in a 3rd-order Indiana stream. Journal of the North American Benthological Society 11:341-349. (i)

Fausch, K.D., and T. G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 49:682-693. (i)

Fetherston, K.L., R.J. Naiman, and R.E. Bilby. 1995. Large woody debris, physical process, and riparian forest development in montane river networks of the Pacific Northwest. Geomorphology 13:133-144. (i)

Fox, M.J. 2001. A new look at the quantities and volumes of instream wood in forested basins within Washington State. Master's thesis. University of Washington, Seattle. (i)

Fox, M.J. 2003. Spatial organization, position, and source characteristics of large woody debris in natural systems. Doctoral dissertation. University of Washington, Seattle. (i)

Fox, M., and S. Bolton. 2007. A regional and geomorphic reference for quantities and volumes of instream wood in unmanaged forested basins of Washington State. North American Journal of Fisheries Management 27:342-359. (i)

Franklin, J.F., and C.T. Dyrness. 1988. Natural vegetation of Oregon and Washington. Oregon State University press, Corvallis, Oregon. (i)

Gregory, K.J. 2003. The limits of wood in world rivers: present, past, and future. Pages 1-19 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Gregory, S.V., K.L. Boyer, and A.M. Gurnell, editors. 2003. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones: focus on links between land and water. Bioscience 41:540-551. (i)

Grizzel, J.D., and N. Wolff. 1998. Occurrence of windthrow in forest buffer strips and its effect on small streams in northwest Washington. Northwest Science 72: 214-223. (i)

Gurnell, A.M. 2003. Wood storage and mobility. Pages 75-92 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Gurnell, A.M., K.J. Gregory, and G.E. Petts. 1995. The role of coarse woody debris in forest aquatic habitats: implications for management. Aquatic Conservation: Marine and Freshwater Ecosystems 5:143-166. (i)

Gurnell, A.M., H. Piégay, F.J. Swanson, and S.V. Gregory. 2002. Large wood and fluvial processes. Freshwater Biology 47:601-619. (i)

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack, and K.W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15:133–302. (i)

Hartman, G.F. 1965. The role of behavior in the ecology and interaction of underyearling Coho Salmon (*Oncorhynchus kisutch*) and steelhead trout (*Salmo gairdneri*). Journal of the Fisheries Board of Canada 22:1035-1081. (i)

Hassan, M.A., D.L. Hogan, S.A. Bird, C.L. May, T. Gomi, and D. Campbell. 2005. Spatial and temporal dynamics of wood in headwater streams of the Pacific Northwest. Journal of the American Water Resources Association 41:899-919. (i)

Henry, W.V., and V.C. Twitty. 1940. Contributions to the life histories of *Dicamptodon ensatus* and *Ambystoma gracile*. Copeia 1940:247-250. (i)

Heede, B.H. 1972. Influences of a forest on the hydraulic geometry of two mountain streams. Water Resources Bulletin 8:523-530. (i)

Hernandez, O., R.W. Merritt, and M.S. Wipfli. 2005. Benthic invertebrate community structure is influenced by forest succession after clearcut logging in southeastern Alaska. Hydrobiologia 533:45-59. (i)

Holland D.C. 1994. The Western Pond Turtle: habitat and history. Bonneville Power Administration, Wildlife Diversity Program, BP-62137-1, unpublished report, Portland, Oregon. (viii)

Hyatt, T.L., and R.J. Naiman. 2001. The residence time of large woody debris in the Queets River, Washington, USA. Ecological Applications 11:191-202. (i)

Hyatt, T.L., T.Z. Waldo, and T.J. Beechie. 2004. A watershed scale assessment of riparian forests, with implications for restoration. Restoration Ecology 12:175-175. (i)

Jackson, C.R., D.P. Batzer, S.S. Cross, S.M. Haggerty, and C.A. Sturm. 2007. Headwater streams and timber harvest: channel, macroinvertebrate, and amphibian response and recovery. Forest Science 53:356-370. (i)

Jamieson, B., and J. Braatne. 2001. Riparian cottonwood ecosystems and regulated flows in Kootenai and Yakima subbasins: impacts of flow regulation on riparian cottonwood forests of the Yakima River. BPA Report DOE/BP-00000005-3. Bonneville Power Administration, Portland, Oregon. (viii)

Johnson, L. B., D.H. Breneman, and C. Richards. 2003. Macroinvertebrate community structure and function associated with large wood in low gradient streams. River Research and Applications 19:199-218. (i)

Johnston, N.T., S.A. Bird, D.L. Hogan, and E.A. MacIsaac. 2011. Mechanisms and source distances for the input of large woody debris to forested streams in British Columbia, Canada. Canadian Journal of Forest Research 41:2231-2246. (i)

Jones, L.L.C., R.B. Bury, and P.S. Corn. 1990. Field observation on the development of a clutch of Pacific Giant Salamander (*Dicamptodon tenebrosus*) eggs. Northwestern Naturalist 71:93-94. (i)

Kauffman, J.B., R.L. Beschta, N. Otting, and D. Lytjen. 1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries 22:12-24. (i)

Keller, E.A., and F.J. Swanson. 1979. Effects of large organic material on channel form and fluvial processes. Earth Surface Processes 4:361-380. (i)

Latterell, J.J., and R.J. Naiman. 2007. Sources and dynamics of large logs in a temperate floodplain river. Ecological Applications 17:1127-1141. (i)

Liquori, M.K. 2006. Post-harvest riparian buffer response: implications for wood recruitment modeling and buffer design. Journal of the American Water Resources Association 42:177-189. (i)

Mack, R.N., and J.N. Thompson 1982. Evolution in steppe with few large, hooved mammals. American Naturalist 119:757-773. (i)

Martin, D.J., and L.E. Benda. 2001. Patterns of instream wood recruitment and transport at the watershed scale. Transactions of the American Fisheries Society 130:940-958. (i)

Martin, D.J., and R.A. Grotefendt. 2007. Stand mortality in buffer strips and the supply of woody debris to streams in Southeast Alaska. Canadian Journal of Forest Research 37:36-49. (i)

Maser, C.M., R.F. Tarrant, J.M. Trappe, and J.F. Franklin, editors. 1988. From the forest to the sea: a story of fallen trees. PNW-GTR-229. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

May, C.L., and R.E. Gresswell. 2003. Large wood recruitment and redistribution in headwater streams in the southern Oregon Coast Range, USA. Canadian Journal of Forest Research 33:1352-1362. (i)

May, C.W., R.R. Horner, J.R. Karr, B.W. Mar, and E.B. Welch. 1997. Effects of urbanization on small streams in the Puget Sound Lowland Ecoregion. Watershed Protection Techniques 2(4):485-494. (i)

McAllister, L.S. 2008. Reconstructing historical riparian conditions for two river basins in eastern Oregon, USA. Environmental Management 42:412-425. (i)

McDade, M.H., F.J. Swanson, W.A. McKee, J.F. Franklin, and J.V. Sickle. 1990. Source distances for coarse woody debris entering small streams in western Oregon and Washington. Canadian Journal of Forest Research 20:326-330. (i)

Minckley, W.L., and J.N. Rune. 1985. Large woody debris in hot-desert streams: an historical overview. Desert Plants 7(3):142-153. (i)

Montgomery, D.R., and J.M. Buffington. 1997. Channel-reach morphology in mountain drainage basins. Geological Society of America Bulletin 109:596–611. (i)

Montgomery, D.R., J.M. Buffington, R.D. Smith, K.M. Schmidt, and G. Pess. 1995. Pool spacing in forest channels. Water Resources Research 31:1097-1105. (i)

Murphy, M.L., J. Heifetz, S.W. Johnson, K.V. Koski, and J.F. Thedinga. 1986. Effects of clear-cut logging with and without buffer strips on juvenile salmonids in Alaskan streams. Canadian Journal of Fisheries and Aquatic Sciences 43:1521-1533. (i)

Murphy, M.L., and K.V. Koski. 1989. Input and depletion of woody debris in Alaska streams and implications for streamside management. North American Journal of Fisheries Management 9:427-436. (i)

Naiman, R.J., E.V. Balian, K.K. Bartz, R.E. Bilby, and J.J. Latterell. 2002a. Dead wood dynamics in stream ecosystems. Pages 23-47 *in* W.F. Laudenslayer, P.J. Shea, B.E. Valentine, C.P Weatherspoon, and T.E. Lisle, technical coordinators. Proceedings of the symposium on the ecology and management of dead wood in western forests. PSW-GTR-181. U.S. Forest Service, Pacific Southwest Research Station, Portland, Oregon. (viii)

Naiman, R.J., R.E. Bilby, and P.A. Bisson. 2000. Riparian ecology and management in the Pacific coastal rain forest. BioScience 50:996-1011. (i)

Naiman, R.J., R.E. Bilby, D.E. Schindler, and J.M. Helfield. 2002b. Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. Ecosystems 5:399–417. (i)

Naiman, R.J., C.A. Johnston, and J.C. Kelley. 1988. Alteration of North American streams by Beaver. BioScience 38:753-762. (i)

Naiman, R.J., and J.R. Sedell. 1979. Characterization of particulate organic matter transported by some Cascade Mountain streams. Journal of the Fisheries Research Board of Canada 36:17-31. (i)

Nakamura, F., and F.J. Swanson. 1993. Effects of coarse woody debris on morphology and sediment storage of a mountain stream system in western Oregon. Earth Surface Processes and Landforms 18:43-61. (i)

Nussbaum, R.A. 1969. Nests and eggs of the Pacific Giant Salamander, *Dicamptodon ensatus* (Eschsacholtz). Herpetologica 25:257–262. (i)

Olson, D.H., P.D. Anderson, C.A. Frissell, H.H. Welsh Jr., and D.F. Bradford. 2007. Biodiversity management approaches for stream–riparian areas: perspectives for Pacific Northwest headwater forests, microclimates, and amphibians. Forest Ecology and Management 246: 81-107. (i)

Parker, M.S. 1994. Feeding ecology of stream-dwelling Pacific Giant Salamander larvae (*Dicamptodon tenebrosus*). Copeia 1994:705-718. (i)

Pereira, C.R.D., and N.H. Anderson. 1982. Observations on the life history and feeding of *Cinygma integrum* (Eaton) and *Ironodes nitidus* (Eaton) (*Ephemeroptera: Heptageniidae*). Melanderia 39:35-45. (i)

Pereira, C.R.D., N.H. Anderson, and T. Dudley. 1982. Gut content analysis of aquatic insects from wood substrates. Melanderia 39:23-33. (i)

Piégay, H. 2003. Dynamics of wood in large rivers. Pages 109-134 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Pollock, M.M., T.J. Beechie, and C.E. Jordan. 2007. Geomorphic changes upstream of Beaver dams in Bridge Creek, an incised stream channel in the interior Columbia River basin, eastern Oregon. Earth Surface Processes and Landforms 32:1174-1185. (i)

Pollock, M.M., M. Heim, and D. Werner. 2003. Hydrological and geomorphic effects of Beaver dams and their influence on fishes. Pages 213-233 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Pollock, M.M., G. Lewallen, K. Woodruff, C.E. Jordan, and J.M. Castro, editors. 2015. The Beaver restoration guidebook: working with Beaver to restore streams, wetlands, and floodplains. Version 1.02. U.S. Fish and Wildlife Service, Portland, Oregon. (viii)

Powell, S.R., L.D. Daniels, and T.A. Jones. 2009. Temporal dynamics of large woody debris in small streams of the Alberta foothills, Canada. Canadian Journal of Forest Research 39:1159-1170. (i)

Ralph, S.C., G.C. Poole, L.L. Conquest, and R.J. Naiman. 1994. Stream channel morphology and woody debris in logged and unlogged basins of western Washington. Canadian Journal of Fisheries and Aquatic Sciences 51:37-51. (i)

Reeves, G.H., K.M. Burnett, and E.V. McGarry. 2003. Sources of large wood in the main stem of a fourth-order watershed in coastal Oregon. Canadian Journal of Forest Research 33:1363-1370. (i)

Reich, M., J.L. Kershner, and R.C. Wildman. 2003. Restoring streams with large wood: a synthesis. Pages 355-366 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Reimchen T.E., D. Mathewson, M.D. Hocking, J. Moran, and D. Harris. 2003. Isotopic evidence for enrichment of salmon-derived nutrients in vegetation, soil, and insects in riparian zones in coastal British Columbia. Pages 59-69 *in* J. Stockner, editor. Nutrients in salmonid ecosystems: sustaining production and biodiversity. American Fisheries Society, Symposium 34, Bethesda, Maryland. (i)

Rhoads, B.L. 1988. Mutual adjustments between process and form in a desert mountain fluvial system. Annals of the Association of American Geographers 78:271-287. (i)

Richardson, J.S. 2008. Aquatic arthropods and forestry: effects of large-scale land use on aquatic systems in Nearctic temperate regions. The Canadian Entomologist 140:495-509. (i)

Robison, E.G., and R.L. Beschta. 1990. Identifying trees in riparian areas that can provide coarse woody debris. Forest Science 36:790-801. (i)

Roni, P., T.J. Beechie, G.R. Pess, and K.M. Hanson. 2014. Wood placement in river restoration: fact, fiction and future direction. Canadian Journal of Fisheries and Aquatic Sciences 72(3):466-478. (i)

Roni, P., and T.P. Quinn. 2001. Density and size of juvenile salmonids in response to placement of large woody debris in western Oregon and Washington streams. Canadian Journal of Fisheries and Aquatic Sciences 58:282–292. (i)

Roy, A.H., C.L. Faust, M.C. Freeman, and J.L. Meyer. 2005. Reach-scale effects of riparian forest cover on urban stream ecosystems. Canadian Journal of Fisheries and Aquatic Sciences 62:2312-2329. (i)

Schuett-Hames, D., A.E. Pleus, J. Ward, M. Fox, and J. Light. 1999. Method manual for the large woody debris survey. TFW-AM9-99-004. Forest Practices Division, Washington Department of Natural Resources, Olympia, Washington. (viii)

Sedell, J.R., P.A. Bisson, F.J. Swanson, and S.V. Gregory. 1988. What we know about large trees that fall into stream and rivers. Pages 47-80 *in* C.M. Maser, R.F. Tarrant, J.M. Trappe, and J.F. Franklin, editors. From the forest to the sea: a story of fallen trees. PNW-GTR-229. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Sedell, J.R., and K.J. Luchessa. 1981. Using the historical record as an aid to salmonid habitat enhancement. Pages 210-233 *in* N.B. Armentrout, editor. Acquisition and utilization of aquatic habitat inventory information. American Fisheries Society, Bethesda, Maryland. (i)

Sedell, J.R., F.J. Swanson, and S.V. Gregory. 1985. Evaluating fish response to woody debris. Pages 222-245 *in* T.J. Hassler, editor. Pacific Northwest stream habitat management workshop. Humboldt State University, Arcata, California. (viii)

Spence, B.C., G.A., Lomnicky, R.M. Hughes, and R.P Novitzki. 1996. An ecosystem approach to salmonid conservation. TR-4501-96-6057. ManTech Environmental Research Services Corporation, Corvallis, Oregon. (viii)

Spies, T., M. Pollock, G. Reeves, and T. Beechie. 2013. Effects of riparian thinning on wood recruitment: a scientific synthesis. Science Review Team Wood Recruitment Subgroup, Forest Science Laboratory, Corvallis, Oregon. (viii)

Steel, E.A., W.H. Richards, and K.A. Kelsay. 2003. Wood and wildlife: benefits of river wood to terrestrial and aquatic vertebrates. Pages 235-248 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Steele, C.A., and C. Brammer. 2006. Dietary overlap in giant salamanders (*Dicamptodon*): applying null models to resource partitioning. Western North American Naturalist 66:115-120. (i)

Stouder, D.J., P.A. Bisson, and R.J. Naiman, editors. 1997. Pacific salmon and their ecosystems: status and future options. Chapman & Hall, New York. (i)

Sullivan, K, T.E. Lisle, C.A. Dolloff, G.E. Grant, and L.M. Reid. 1987. Stream channels: the link between forests and fishes. Pages 30-97 *in* E.O. Salo and T.W. Cundy, editors. Streamside management: forestry and fishery interactions. Contribution No. 57. Institute of Forest Resources, University of Washington, Seattle. (viii)

Swanson, F.J., G.W. Lienkaemper, and J.R. Sedell. 1976. History, physical effects, and management implications of large organic debris in western Oregon streams. PNW-GTR-56. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Tschaplinski, P.J., and G.F. Hartman. 1983. Winter distribution of juvenile Coho Salmon (*Oncorhynchus kisutch*) before and after logging in Carnation Creek, British Columbia, and some implications for overwinter survival. Canadian Journal of Fisheries and Aquatic Sciences 40:452-461. (i)

Van Sickle, J., and S.V. Gregory. 1990. Modeling inputs of large woody debris from falling trees. Canadian Journal Forest Research 20:1593-1601. (i)

Ward, G.H., K.W. Cummins, R.W. Speaker, A.K. Ward, S.V. Gregory, and T.L. Dudley. 1982. Habitat and food resources for invertebrate communities in South Fork Hoh River, Olympic National Park. Pages 9-14 *in* E.E. Starkey, J.F. Franklin, and J.W. Matthews, editors. Ecological research in national parks of the Pacific Northwest. Forest Research Laboratory, Oregon State University, Corvallis, Oregon. (viii)

WDNR (Washington Department of Natural Resources). 2005. Forest practices habitat conservation plan. Forest Practices Division, Washington Department of Natural Resources, Olympia, Washington. (v)

Welsh, H.H., Jr., and A.J. Lind. 2002. Multiscale habitat relationships of stream amphibians in the Klamath-Siskiyou Region of California and Oregon. Journal of Wildlife Management 66:581-602. (i)

WFPB (Washington Forest Practices Board). 1987. Proposed forest practices rule and regulations: final environmental impact statement. Washington Forest Practices Board, Washington Department of Natural Resources, Olympia, Washington. (v)

WFPB (Washington Forest Practices Board). 2011. Board manual: standard methodology for conducting watershed analysis. Version 5.0. Appendix F: Fish habitat module. Washington Forest Practices Board, Washington Department of Natural Resources, Olympia, Washington. (v)

Wiggins, G.B. 1977. Larvae of North American Caddisfly Genera (Trichoptera). 1st Edition. University of Toronto Press, Ontario, Canada. (i)

Wilkins, R.P., and N.P. Peterson. 2000. Factors related to amphibian occurrence and abundance in headwater streams draining second-growth Douglas-fir forests in southwestern Washington. Forest Ecology and Management 139:79-79. (i)

Wing, M.G., and A. Skaugset. 2002. Relationships of channel characteristics, land ownership, and land use patterns to large woody debris in western Oregon streams. Canadian Journal of Fisheries and Aquatic Sciences 59:796-807. (i)

Wisseman, R.W., and N.H. Anderson. 1984. Mortality factors affecting trichoptera eggs and pupae in an Oregon coast range watershed. Pages 455-460 *in* J.C. Morse, editor. Proceedings of the 4th International Symposium on Trichoptera. Dr W. Junk Publishers, The Hague, Netherlands. (viii)

Wissmar, R.C. 2004. Riparian corridors of eastern Oregon and Washington: functions and sustainability along lowland-arid to mountain gradients. Aquatic Sciences 66:373-387. (i)

Wissmar, R.C., J.E. Smith, B.A. McIntosh, H.W. Li, and G.H. Reeves. 1994. A history of resource use and disturbance in riverine basins of eastern Oregon and Washington (early 1800s-1990s). Northwest Science 68 (special issue):1-35. (i)

Wondzell, S.M., and P.A. Bisson. 2003. Influence of wood on aquatic biodiversity. Pages 249-264 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Zalewski, M., M. Lapinska, and P.B. Bayley. 2003. Fish relationships with wood in large rivers. Pages 194-212 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland. (i)

Chapter 4. Stream Temperature

By: Kirk L. Krueger, Daniel J. Isaak, and Marc P. Hayes

4.1. Introduction

The great diversity of Washington's landscapes creates equally diverse stream thermal conditions (Figure 4.1). Broad ranges in elevation (0 -14,000 ft, 0 - 4,300 m), precipitation, stream size, topography, and other factors contribute to some streams having average summer temperatures as low as 39°F (4°C); whereas others only a few miles away may exceed 68°F (20°C). Much of that thermal heterogeneity is dictated by effectively immutable geomorphic attributes of landscapes (e.g., elevation, aspect, topographic slope), which affect the potential amount of solar radiation available at a given location (Boyd and Kasper 2003). However, the types of riparian vegetation and their condition also play important roles in determining the amount of solar radiation that reaches a stream's surface. Through management of riparian ecosystem conditions, especially vegetation, the spatiotemporal distribution of stream temperatures (i.e., thermal regime) (Boyd and Kasper 2003; Caissie 2006) can be affected, which in turn, directly and indirectly affect the survival and productivity of aquatic species (Beschta et al. 1987; McCullough 1999) including salmon.

Shade can substantially reduce the amount of direct (shortwave) solar radiation, usually the main cause of heating, that reaches a stream (Poole and Berman 2001). Indirect effects of riparian vegetation include, but are not limited to, maintenance of channel form and

Pink Salmon Oncorhynchus gorbuscha *moving upstream to spawn/Ned Pittman, WDFW* hyporheic connectivity (Gregory et al. 1991; Stanford 1998; Poole and Berman 2001) that also affect stream temperatures (Boyd and Kasper 2003; Webb et al. 2008). Maintaining and restoring riparian ecosystem functions and the thermal regimes of aquatic systems is increasingly important in light of climate change and as land use intensifies. Land use can strongly influence the intensity, timing, duration, and geographic distributions (i.e., structure) of stream temperatures (NRC 2002).

In this chapter we briefly summarize the results of our literature review, and present a simple conceptual model that identifies components (environmental factors) and structures (relations of components) that affect stream temperatures, to facilitate understanding of potential effects of different management alternatives. We also briefly discuss considerations of the effects of riparian management on stream thermal regime within a watershed or stream network context, the sensitivity of some priority species to changes in stream thermal regime, and applications of current scientific information to riparian management for suitable thermal regimes for aquatic species. Much of the literature describing the effects of human activity on stream temperature has focused on summer maximum water temperatures. Increasingly, scientists are focusing on better understanding stream thermal regimes, which describe how the temperature of streams varies through time and throughout stream networks. This shift recognizes two important ideas: 1) fish and other aquatic organism use different stream habitats, defined in part by specific water temperature ranges, to complete their life cycle, and 2) heat energy inputs into streams are variable in space and time and heat energy is often carried downstream of the reach from which it was gained.

Our review of the scientific literature on the effects of riparian conditions on stream temperatures was extensive; including more than 6,000 articles relevant to riparian ecosystem research, of which more than 700 referenced stream temperatures and over 100 that referred to or reported measured effects of riparian ecosystem conditions on stream temperatures. Our work also identifies research to inform management recommendations that are applicable across a wide range of environmental conditions (e.g., various latitudes, topographies, and land uses). Further, we wanted to address the broad range of thermal conditions that are important to aquatic species (e.g., frequency, duration, intensity, and predictability of summer high temperatures and temperatures during incubation (Lytle and Poff 2004); and the geographic distribution of stream temperatures (Torgersen et al. 1999; Poole et al. 2001; Ebersole et al. 2003a). Therefore, we reference additional scientific information to provide the context that allows for better understanding and application of riparian ecosystem-specific information. Several reviews were particularly valuable in directing our efforts (e.g., Elmore and Kauffman 1994; May 2003; Moore et al. 2005; Sather and May 2007; Bowler et al. 2012; Czarnomski and Hale 2013).

> *Scientists are focusing on better understanding stream thermal regimes, which describe how the temperature of streams varies through time and throughout stream networks.*

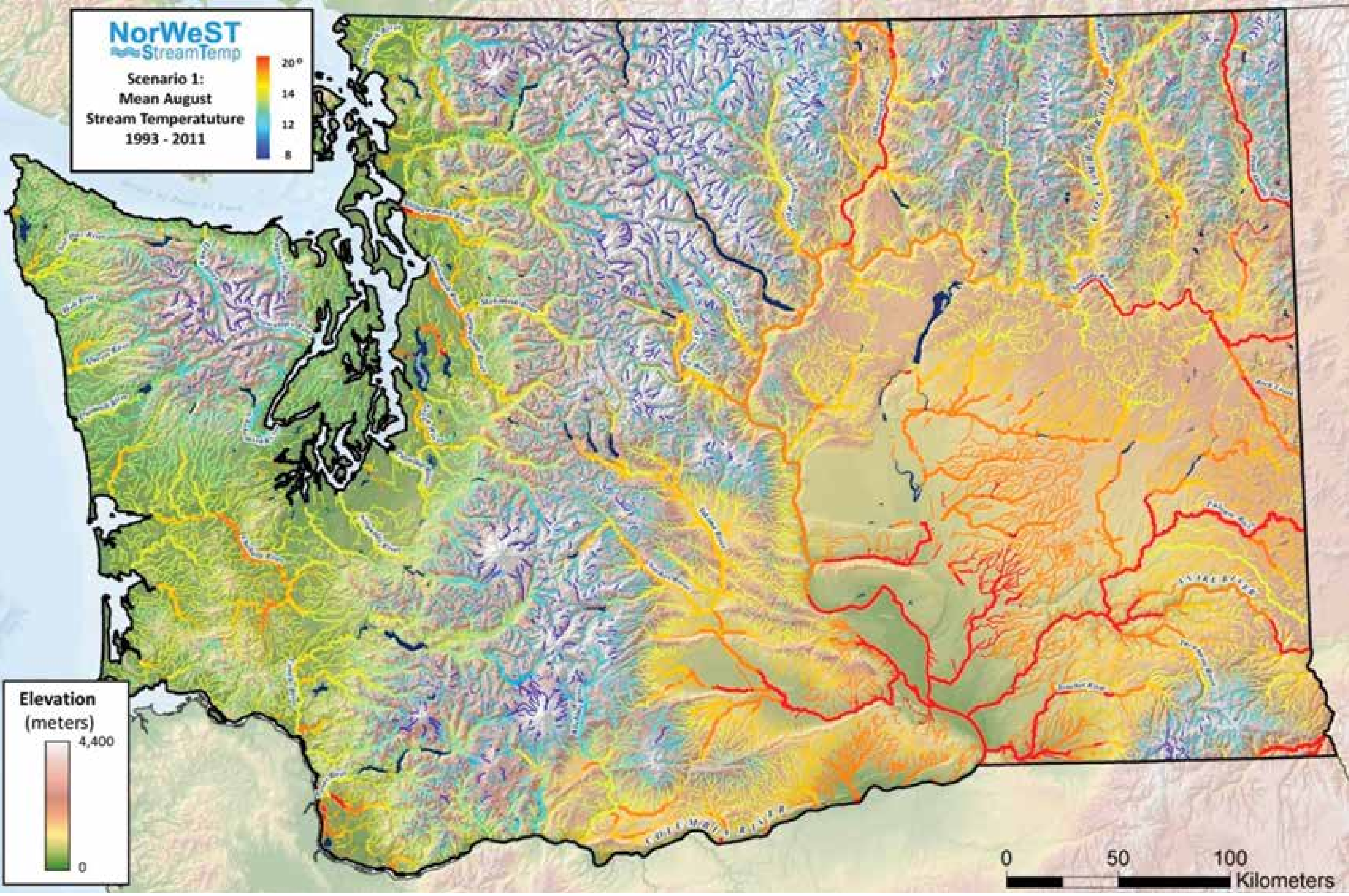

Figure 4.1. Stream thermalscape showing mean August temperatures for 66,236 kilometers (41,157 miles) of streams across Washington that was developed in the NorWeST project (poster available here: http://www.fs.fed .us/rm/boise/AWAE/projects/stream_temp/downloads/14NorWeST_WashingtonStreamTemperatureMap.pdf). Topographic and geomorphic complexity across the state creates significant thermal heterogeneity that is moderated locally by riparian conditions.

4.1.1. State of the Science

The scientific literature establishes a clear expectation that reduction of stream shade, especially due to vegetation removal, will result in warmer summer stream temperatures (e.g., Sridhar et al. 2004; Allen et al. 2007). The literature, derived primarily from descriptive (e.g., case study) and occasionally statistically predictive (e.g., generalization across many case studies) studies, demonstrates that riparian management can affect shade and in turn stream temperatures (Johnson 2004; Moore et al. 2005). The vast majority of those studies document an increase in a temperature statistic, often mean daily maximum temperature during summer, associated with a loss of shade (Johnson 2004; Moore et al. 2005; Bowler et al. 2012), but a few studies have shown mixed results (Johnson 2004; Moore et al. 2005; Janisch et al. 2012; Kibler et al. 2013).

Variability in stream temperatures among locations (Figure 4.1) and among some study results is due to both temporally constant attributes of landscapes (e.g., elevation, topographic slope, drainage area) and temporally variable attributes (e.g., stream wetted width, stream flow) that affect the potential amount of solar radiation available at a given location (Boyd and Kasper 2003) and the thermal sensitivity of the stream, respectively. The amount of solar radiation available is often referred to as thermal loading potential or potential solar load. At a given thermal input or load, the temperature change responsiveness of a stream is often referred to as "thermal sensitivity" (Beschta et al. 1987; Moore et al. 2005). Thermal sensitivity is influenced by many physical factors. Streams with higher flows of surface and groundwater, and hyporheic exchange rates (Cristea and Burges 2010; Arismendi et al. 2012) tend to be relatively thermally insensitive. Wood loading, channel complexity and depth can also affect thermal load and sensitivity. The potential solar load in combination with thermal sensitivity (discussed in detail below) of locations across a stream network and through time largely determine a stream thermal regime.

In addition to solar radiation, other microclimate variables - such as air temperature, wind speed, and humidity - may affect instream water temperatures. Microclimate conditions interact with energy exchanges at the air-water interface (Figure 4.2) and can thereby affect stream thermal regimes (Moore et al. 2005; Benyahya et al. 2010), especially in small streams (Caissie 2016). However, we believe the effects of microclimate on the thermal regime of streams with fully functioning riparian ecosystems are minor for two reasons: 1) microclimate effects (e.g., changes in air temperature and humidity) at forest edges rarely extend farther than one tree height into mature riparian forest (Moore et al. 2005; Rykken et al. 2007; Reeves at al. 2018), and 2) sensible heat exchanges comprise only a small portion of total heat flux in streams (Johnson 2004; Moore et al. 2005). In fact, net solar radiation effects on stream temperatures are generally about an order of magnitude greater than sensible and latent heat exchanges at the air-water interface (Moore et al. 2005; D. Caissie, Fisheries and Oceans Canada, personal communication). However, we also agree with Reeves et al. (2018), who note that the range of effects measured in different studies suggests substantial uncertainties regarding riparian ecosystem management with respect to microclimate.

Riparian ecosystem-stream temperature research appears to be transitioning from largely local, descriptive (e.g., case study and generalizations of case studies) to predictive, mechanistic, and deterministic modeling across larger areas (e.g., Boyd and Kasper 2003; Allen et al. 2007). Mechanistic models attempt to include most of the factors that affect stream temperatures (Figure 4.2) in a given area and thus do not rely on statistical extrapolation from other locations. These types of models can provide relatively accurate and precise temperature predictions at various spatial and temporal scales (Leinenbach et al. 2013). Further, they can provide useful information on systems and processes that are not readily estimated through statistical approaches used in case studies. However, collecting the environmental data necessary for accurate prediction of the effects of riparian management actions on stream thermal regimes via mechanistic models is difficult and often prohibitively expensive (Allen et al. 2007). For example, studies on the effects of riparian ecosystem or land cover management on stream temperatures at large spatial extents (e.g., watersheds of large streams or rivers) and cumulative effects are very rare. Watershed-scale studies (e.g., Janisch et al. 2012; Kibler et al. 2013) are often limited to relatively small watersheds (e.g., 2 to 1,000 ha) and second order streams.

New types of statistical models for stream network data (Ver Hoef et al. 2006; Isaak et al. 2014) have been applied to large stream temperature databases (Isaak et al. 2017a) and these have enabled relatively accurate predictions ($R^2 = 0.90$; RMSE = 1.8°F or 1.0°C) and mapping of summer thermal conditions across all Washington streams (Figure 4.1). Those statistical approaches use predictor variables derived from broad geospatial and remotely sensed datasets (e.g., elevation, stream slope, riparian density from the National Land Cover Database) so they do not provide insights to local temperature anomalies that could be associated with alterations of riparian or channel conditions. However, these statistical models do provide accurate thermal status maps that could be used with mechanistic models or field measurements to better ascertain the influence of riparian conditions on stream temperatures.

Scientific knowledge of the importance of stream temperatures on aquatic species has also advanced since the publication of Knutson and Naef in 1997; especially by better describing the importance of thermal regimes (rather than only extreme high temperatures) on fish survival, growth, and productivity (Ward and Stanford 1982; McCullough 1999; Caissie 2006; McCullough et al. 2009; Hinch et al. 2012). For example, the collapse of the Fraser River Sockeye Salmon *Oncorhynchus nerka* spawning run in 2009 was largely attributed to thermal exposure due to early migration, which resulted in indirect, but often fatal, effects (Hinch et al. 2012). Following Olden and Naiman (2010), we suggest that thermal regimes can be described by their rates of change, magnitude, frequency, timing, and duration, and also their spatial distribution in a stream. The timing and duration of thermal exposure, in addition to exposure to extreme temperatures, affects the survival and productivity of many aquatic species (Caissie 2006).

Importantly, studies such as those of Murphy and Hall (1981), Murphy et al. (1981), Hawkins et al. (1983), and Bilby and Bisson (1987) that demonstrated apparent beneficial effects of reduced riparian shade on an aquatic species, such as increased salmonid growth or abundance, can now be considered in a broader ecological context. It should be noted that in these studies observed temperature increases were small and apparent beneficial effects were attributed to increased solar radiation, primary production, and consequent bottom-up stimulation of the food web. These findings are consistent with the idea that oftentimes the biological effects of altered temperatures differ among locations (Li et al. 1994; Farrell et al. 2008), and that such differences in effects may be attributed to the difference between local temperature and the thermal niche of a species. For example, in reaches where temperatures are near a species' thermal maxima, additional warming will cause the habitat to become unsuitable. However, in streams that are too cold for a species, warmer temperatures may increase habitat suitability (Isaak and Hubert 2004; Isaak et al. 2017b). In addition to direct stream temperature changes on a single species, changes may affect ecological trade-offs between species. Rezende et al. (2014) showed that faster growth might also be associated with increased competition with non-native species (Lawrence et al. 2014).

Widespread alterations of upland and riparian conditions and subsequent changes in stream composition and structure have likely already impacted thermal conditions in many streams (Poole and Berman 2001) and such impacts will be further exacerbated by the ongoing effects of climate change (Isaak et al. 2012; Holsinger et al. 2014). Streams with degraded riparian conditions, however, also offer opportunities where restoration actions could mitigate future warming and improve ecological resilience by enhancing the survival and productivity of the populations and assemblages that use those systems.

4.2. Conceptual Model

We provide a conceptual model to help describe the important components, structures, and processes that affect stream temperatures and that can result in different effects of riparian management actions on stream temperatures among locations (Figure 4.2). Our conceptual model is largely a simplification of the model "Heat Source" in Boyd (1996) and Boyd and Kasper (2003) and developed based on the preponderance of scientific evidence. Heat Source is a deterministic, mechanistic model developed to predict dynamic open channel heat and mass transfer. As such, it includes parameters for all variables that are known to significantly affect stream temperatures. Our simplified conceptual model describes components and parameters that can have a large effect on stream temperatures or that are affected by riparian ecosystem management. We provide this model in an attempt to improve basic understanding of stream thermal processes and to identify important composition, structure, and functions that are subject to management and that affect stream temperatures. Additional information based on the models of Poole and Berman (2001), Moore et al. (2005), and Leinenbach et al. (2013) are included to more clearly identify factors that affect thermal loading potential and stream thermal sensitivity via riparian management.

Scientific understanding of the environmental factors and their relations that affect stream thermal regimes is well developed, as demonstrated by the development of deterministic, mechanistic models (e.g., Boyd and Kasper 2003) that can provide accurate temperature predictions when sufficient data are available to run the model and when the stream system remains relatively stable. Such models have been applied successfully at the spatial extent of individual stream reaches (Cristea and Janisch 2007; Cole and Newton 2013) and watersheds (Coffin et al. 2011; Bisson et al. 2013; Booth et al. 2014), suggesting that our understanding of the composition, structure, and functions that affect stream temperatures are well understood and can be

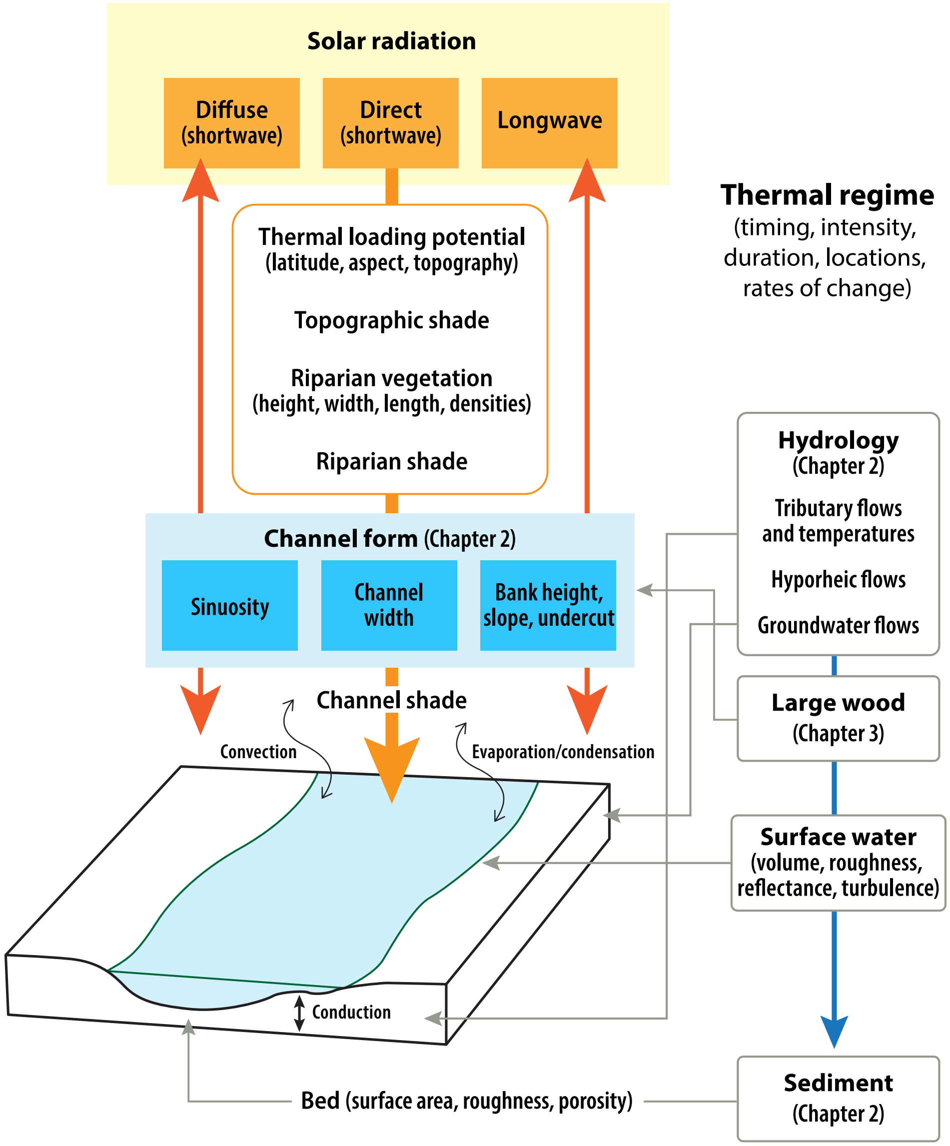

Figure 4.2. Conceptual model that identifies some factors that affect the thermal regime of stream networks. Relative importance is noted by line weight and arrows indicate directions of flows. Note that most of the elements in the model are influenced by human activities and that many components and structures are not presented here. See Boyd and Kasper (2003) for a more thorough review.


Dense tree canopies rarely provide 100% shade/Ned Pittman, WDFW

reliably applied to management problems. Rates of direct (shortwave) solar radiation reaching the stream surface and the volumes and temperatures of water are consistently identified as the dominant processes affecting the thermal regime of streams (Sinokrot and Stefan 1993; Johnson 2004). At many locations these two dominant processes are subject to management: direct solar radiation reaching a stream via vegetation management and water flows via water diversion, storage, and return (Olden and Naiman 2010).

The thermal loading potential of a stream reach is controlled in part by temporally constant and largely independent factors such as latitude, aspect, elevation, and topographic shading (Figure 4.2, Boyd and Kasper 2003; Janisch et al. 2012). Streams at high latitudes are less exposed to solar radiation, the ultimate source of heat energy to streams, than streams at low latitudes. Stream aspect affects stream temperatures because a stream's orientation relative to the path of the sun (which differs seasonally) determines the amount of direct solar radiation that a stream receives (Johnson 1971). Elevation affects the ambient air temperatures that streams are exposed to, as well as the dominant form and amount of precipitation (i.e., snow or rain). For these reasons, higher elevation streams are usually colder and commonly, although not always, less thermally sensitive than lower elevation streams (Luce et al. 2014). Topography can shade a stream from direct solar radiation and the effects of topography and aspect can interact. The effects of these four geographic factors can be difficult to detect (Isaak and Hubert 2001) and are likely responsible for some of the variability of the results of case studies that measured the effects of riparian management on stream temperatures. Note that some of these attributes (e.g., elevation) also affect the composition and structure of riparian vegetation. Stream thermal loading is not only controlled by independent factors such as latitude, aspect, elevation, and topographic shading. The scientific literature clearly identifies shade from riparian vegetation as important to stream temperatures, especially for small (narrow) streams (e.g., Beschta 1987; Poole and Berman 2001; Johnson 2004; Sridhar et al. 2004; Moore et al. 2005; Caissie 2006; Allen et al. 2007; Bowler et al. 2012; Garner et al. 2014). The effects of altered stream shading on temperatures is complex because thermal loading potential and thermal sensitivity of streams differ among locations. Some attributes, such as water volume, wetted width, and width-depth ratio can affect both thermal loading potential and thermal sensitivity (Cristea and Janisch 2007; DeWalle 2010). For a given volume of water, wider channels of the same length have a greater surface area that can intercept more direct solar radiation (Boyd and Kasper 2003) and also exchange more heat through atmospheric conduction. The width of a stream also affects the amount of shade that can be provided by topography and riparian vegetation (Cristea and Janisch 2007; DeWalle 2010). That is, riparian vegetation of given height, density, and width might shade a larger proportion of a narrow channel for a longer period than for a wide channel. Stream widths, especially wetted channel widths, are often much less temporally stable than the other factors that control thermal loading potential. Management actions that result in changes in stream width (e.g., over-grazing [Belsky et al. 1999; Chapter 2] and maintenance of wide forested riparian areas that can result in wide channels [Sweeney and Newbold 2014]) can increase thermal loading potential and thermal sensitivity.

While shortwave radiation input and water volume and temperature largely control temperature regime, longitudinal channel form (stream reach and cross section) can also affect stream temperature, for example, via high sinuosity and the presence of undercut banks (Frissell et al. 1986; Bisson et al. 2006). Channel form and features that increase stream roughness can store gravel (Buffington et al. 2004) and increase hyporheic exchange (Wondzell and Gooseff 2013) that, in turn, can affect stream temperatures by storing and releasing heat (Burkholder et al. 2008). Large wood can also store large amounts of gravel in some streams (Buffington et al. 2004) and affect channel form (Chapter 3; Keller and Swanson 1979; Abbe and Montgomery 1996; Konrad et al. 2005) which might subsequently affect stream temperatures. For example, sediment deposition due to instream large wood might increase hyporheic flow and stream width, which can subsequently effect stream temperatures. Steep or undercut banks can also provide shade (Boyd and Kasper 2003) which, in some locations (Ebersole et al. 2003b), can lower water temperatures at small spatial extents (e.g., individual pools; Ebersole et al. 2003a), even in large streams and rivers. Unfortunately, relatively few studies quantify the effects of channel form on thermal sensitivity and stream temperatures, which likely adds variability among stream temperature case studies and limits our ability to predict their relative importance at specific locations. Groundwater and tributary flows also affect stream temperatures (Malard et al. 2002) in two ways, 1) by adding volume and increasing flow velocity that can make streams less sensitive to thermal change (i.e., increasing thermal capacity), and 2) by adding water that is at a different temperature than receiving waters (Boyd and Kasper 2003; Story et al. 2003). However, the effects of groundwater flows can be difficult to detect due to spatiotemporal lags (Alvarez et al. 2004; Arrigoni et al. 2008) and difficult to apply to management due to the paucity of information describing groundwater flows. Additionally, side channels and wetlands can provide large volumes of water at different temperatures to receiving stream water (Bobba et al. 2010). Streambed attributes, such as porosity and surface area, are largely controlled by the flows of water and available sediment (see Chapter 2). Streambed attributes can affect the thermal regime of a reach by affecting heat conduction (Sinokrot and Stefan 1993), by regulating the amount and velocity of water flow through the streambed, and by controlling the rates and locations of groundwater flows (Malcolm et al. 2002). The effects of these processes on stream temperatures can be spatially and temporally discontinuous and dynamic, especially in systems where stream form is heterogeneous and dynamic (Wright et al. 2005; Wondzell 2012). Hyporheic flows often have little effect on average stream surface water temperatures except for small streams at low flows but can create discrete locations of cool water (Wondzell 2012) that may serve as important refugia for salmon. Hyporheic flows can be influenced by flow regulation (e.g., dams), which can reduce variation in both flows and temperatures that reduces the potential for hyporheic exchange to act as a thermal buffer (Ward and Stanford 1995; Poole and Berman 2001; Nyberg et al. 2008).

Juvenile salmon/Portia Leigh, WDFW

The interaction of complex in- and near-stream riparian ecosystem processes can lead to a cascade of events (i.e., a series of related events triggered by single event) and new system states (Pringle and Triska 2000; Allan 2004). These cascading events are almost impossible to predict. For example, the removal of riparian trees typically reduces shading and increases solar radiation to the stream. Even when reductions in shade have a small direct effect on stream temperatures, the loss of trees can increase bank susceptibility to erosion (Chapters 2 and 3; Micheli and Kirchner 2002; Gomi et al. 2004) resulting in a wider stream that increases thermal loading and thermally sensitivity of the stream. Similarly, channel incision resulting from riparian management, may increase local topographic shading to the stream, but lower streambed and water elevations that can shift the riparian vegetation composition and structure. This shift in riparian vegetation can result in less stable streambanks and less riparian shade that ultimately offsets any benefits of topographical shading on water temperature (Toledo and Kauffman 2001). The results of such interactions are difficult to predict because they rely on accurate predictions of changes to stream morphology, vegetation composition and structure (which interact and have non-linear responses), and water and sediment flows (which can be stochastic).

The processes that affect the timing, duration, and geographic distribution of stream temperatures are complex due to the variability of the environment in which these processes occur (Figure 4.2). The relative importance of these ecological processes differs among locations and through time (Poole and Berman 2001; Webb et al. 2008). Scientific literature is clear about the expected effect of reducing stream shading, especially during summer; water gets warmer, particularly peak temperatures. Science is less able to provide accurate and precise predictions of the effects of specific riparian management measures. For example, the studies presented in Figure 4.3 represent statistical models relating changes in riparian buffer width to changes in stream temperature. Data in Figure 4.3 are collected by measuring stream temperature before and after a reduction in the width of a forested riparian buffer which in turn decreases shade to the stream. Model output (prediction) is change in temperature related to specific buffer widths. The models tell us that reducing shade usually increases stream temperatures (following the shape of the dotted line connecting points), but they cannot predict exactly how much shade reduction will result in a specific temperature increase. A manager attempting to use the statistical model in Figure 4.3 to predict how a reduction from a fully forested site to a 30-foot wide buffer might affect summer stream temperatures is confronted with a range of possible answers. Those answers fall within the 90% credible intervals for one study representing a change between 1.25 and 2.25°C (2.25 and 4.05°F), and for another study between -0.5 to nearly 5.0°C (-0.9 to nearly 9.0°F). Uncertainty in this case describes our inability to accurately predict exact outcomes. While rarely applied due to their expense, mechanistic models and attendant data (e.g., hyporheic flows and temperatures) provide the best approach for predicting accurate site-specific outcomes from management.

Fortunately, the literature is very clear that solar radiation is often the dominant factor affecting stream heat budgets during all seasons except winter (Webb et al. 2008). Results of case studies can prove very useful by clearly describing the expected direction and shape of the relations between attributes of riparian ecosystems (e.g., width of vegetation) and stream shade and temperature. Importantly, riparian shade is often directly amenable to management and in many locations riparian vegetation height and density may be more closely associated with shade than width of riparian vegetation per se (DeWalle 2010). In any case, the literature is very clear about the importance of riparian vegetation for providing shade to streams (Figure 4.3a).

4.3. Species Sensitivity

4.3.1. Fish

The scientific literature describing the importance of temperature, especially high temperatures, to fish is extensive (e.g., Brett 1952, 1971; Brett et al. 1969; Richter and Kolmes 2005). As aquatic ectotherms, the physiology of fish are strongly affected by water temperatures (McCullough et al. 2001; Welsh et al. 2001). Studies have frequently found associations between temperature and the geographic distribution (Welsh et al. 2001), spawn timing (Hodgson and Quinn 2002), growth rates, egg development and survival, competitive interactions, life stage survival, and behavior of fish (McCullough 1999; McCullough et al. 2009). Salmonids have frequently been studied because of their cultural and economic importance and their relative sensitivity to high temperatures, narrow thermal tolerance, and narrow aerobic scope (Farrell et al. 2008; Eliason et al. 2011; Ayllon et al. 2013). Summer temperature information is useful for identifying acute problems, but may be insufficient for ensuring population resilience because indirect effects and exposure to altered thermal regimes, not just extreme summer temperatures, can affect fish survival and productivity (McCullough 1999).

Recent research has increasingly focused on the effects of water temperatures and thermal regimes for fishes that have narrow thermal tolerance (i.e., range of suitable temperatures) or aerobic scope (Farrell et al. 2008; Eliason et al. 2011; Ayllon et al. 2013). This work is particularly relevant to salmonids in the Pacific Northwest that have life histories adapted to historical, location-specific thermal regimes (Brannon et al. 2004; Farrell et al. 2008; Eliason et al. 2011). Much of this research is related to the increasing threats posed by climate change, and some studies have already found evidence of existing, detrimental effects of increased stream temperatures on fish. For example, Isaak et al.

Juvenile anadromous salmon can spend up to 3 years in streams depending on the species/Ned Pittman, WDFW

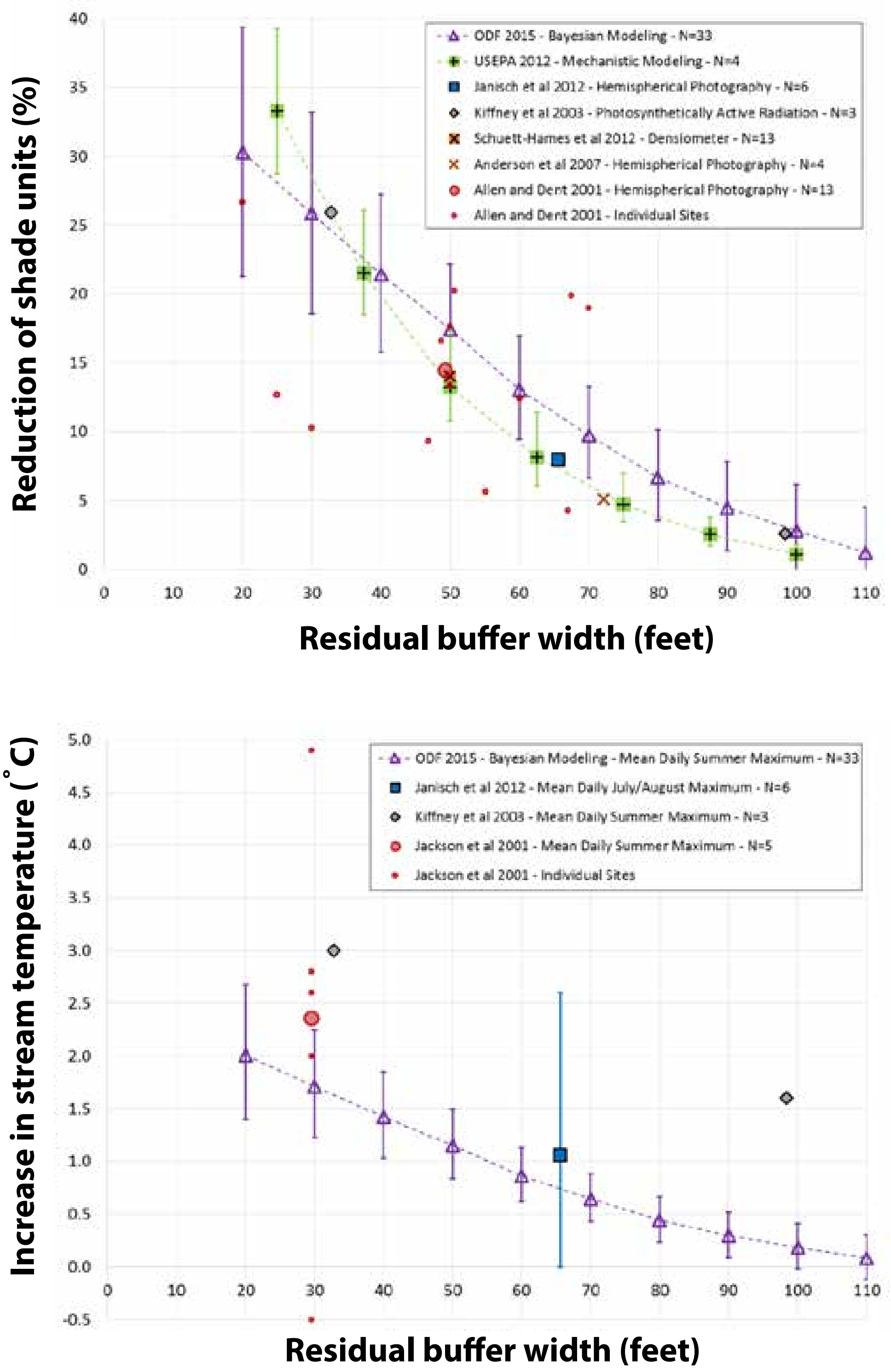

Figure 4.3. Observed shade (a: top panel) and temperature response (b: bottom panel) associated with "no-cut" riparian buffers with adjacent clearcut harvest. Only studies that employed a Before-After-Control-Impact design and conducted in Pacific Northwest forests are included. Shade unit is percent from completely shaded. Bayesian modeling results (and 90% credible intervals) were derived from data collected as part of Groom et al. (2011). Analyses provided by P. Leinenbach, USEPA Region 10.

Chum Salmon Oncorhynchus keta *in spawning colors/Ned Pittman, WDFW*

(2010, 2015) found increases in stream temperatures would result in losses of Bull Trout *Salvelinus confluentus* and Cutthroat Trout *Oncorhynchus clarkii* habitats, but also that especially cold streams could serve as climate refugia during the 21st century. Additionally, a number of recent studies have described some complex effects of altered thermal regimes to the productivity and survival of fish at different life stages and dramatic differences in thermal tolerance and aerobic scope among populations and species. For example, Hinch et al. (2012) describe complex links between the physiology, behavior, environment, disease exposure, and the effect of thermal exposure, on the population-specific survival of migrating Sockeye Salmon. Hodgson and Quinn (2002), Keefer et al. (2007), Strange (2010) and others have documented that high temperatures and previous thermal exposure can inhibit fish migration, and Marsh (1985) and Kappenman et al. (2013) have documented that high and low temperatures can impede spawning and embryo development and that optimal temperatures for embryo survival and development differ among species. Steel et al. (2012) found that thermal variability (without changing mean temperature) can affect the emergence timing and development of Chinook Salmon *Oncorhynchus tshawytscha*.

The literature is conclusive about the importance of stream thermal regimes to fish survival and productivity and also demonstrates wide variability in sensitivity to altered thermal regimes among species and among populations within species (Steel et al. 2012). More research would help us fill the sizeable gap in our understanding of the risk posed by altered thermal regimes on the survival and productivity of populations of salmonids or non-salmonid fishes, and on the ecological interactions that affect survival, productivity and population viability. For example, the time of hatching (i.e., duration of incubation) of salmonid eggs is largely determined by their thermal exposure during incubation. Relatively small changes in temperature can advance or delay maturation and subsequently hatching date (Groot and Margolis 2010) and effects of temperature can differ among closely related populations of the same species that spawn in different habitats (Hendry et al. 1998). Regardless of the ultimate cause, changes in thermal regimes can, for example, uncouple trophic interactions due to temporal mismatches, such as fry emerging when their preferred food is not available to them (Winder and Schindler 2004; Post et al. 2008). Altered thermal regimes can also affect other life stages by altering the timing of parr-smolt transformation (Zaugg and Wagner 1973), the duration of freshwater rearing (Sauter et al. 2001) which can affect ocean survival (Weitkamp et al. 2015), and the timing and success of spawning migration (Crossin et al. 2008). Altered thermal regimes can also affect ecological interactions. For example,

Wise management of riparian and aquatic systems would ideally consider the distribution and dynamics of stream temperatures in space and time and at scales that are relevant to the species in the region.

Lawrence et al. (2014) demonstrated that higher stream temperatures can expand the geographic distribution of non-native Smallmouth Bass *Micropterus dolomieu* and increase their predation on salmon. These results are complicated by the fact that effects of altered thermal regimes often differ among populations and life stages of a species, and among species (Brett 1971; Zaugg and Wagner 1973) which makes prediction of the exact effects of any riparian shade reduction and subsequent effects on stream thermal regime on fish survival and productivity uncertain. This uncertainty is compounded by fish behavioral responses, such as daily and seasonal movement among locations where thermal conditions differ (Torgersen et al. 2001).

Our understanding of the importance of altered thermal regimes on fish has improved substantially since the publication of Knutson and Naef (1997). Discussion of stream temperature by Knutson and Naef (1997) focused mostly on summer high temperatures and their effects on fish, but did not identify many components and structures of ecosystems that can affect stream temperatures or the importance of thermal regimes to fish. In particular, more recent studies are able to measure thermal exposure of fish through time, occasionally in field conditions, and estimate physiological attributes such as growth rate (Sauter and Connolly 2010).

Most studies calculate a single statistic that describes an important attribute of a stream's thermal regime (e.g., mean maximum daily summer temperature) and identify species, population, or life stage attributes that are correlated with that statistic, such as a critical thermal limit. These studies have been useful for demonstrating the importance of temperature and for setting some management objectives. However, they are insufficient for effective, long-term management of perturbed systems because changes in temperature means (i.e., chronic exposure) or extremes (i.e., acute exposure) can result in different selective pressures and evolutionary responses (Rezende et al. 2014) and because life stages, populations, and species differ widely in their sensitivity to altered thermal regimes (Farrell et al. 2008; Hinch et al. 2012). Further, we now better understand that many individual fish and especially populations use (and perhaps are dependent on) a dynamic mosaic of habitats (Fausch et al. 2002; Ward et al. 2002; Wiens 2002); that is, they move between discrete locations with different temperatures (Torgersen et al. 1999). Such movement among "patches" can be important to individual survival and population persistence because thermal conditions at discrete locations can be highly temporally variable (Dugdale et al. 2013) and spatially complex (Fullerton et al. 2015). Management of stream thermal regimes should carefully consider the spatiotemporal distribution and dynamics of thermal conditions at scales that are relevant to the species in the region (Fausch et al. 2002) at durations that are relevant to the rates of system change and recovery from disturbance.

4.3.2. Amphibians and Reptiles

Few studies address the effects of altered thermal regimes on stream-associated amphibians and reptiles, which currently limits the level of specificity to which reliable management recommendations can be made. However, some important insights are possible and these are best discussed in the context of breeding habitat: 1) amphibians that breed in the flowing portions of stream networks, and 2) amphibians and reptiles that breed in or use off-channel stillwater habitats associated with stream networks.

Instream-breeding amphibians (giant salamanders, *Dicamptodon* spp.; tailed frogs, *Ascaphus* spp.; and torrent salamanders, *Rhyacotriton* spp.) are generally described as having low thermal requirements (Bury 2008). However, those requirements are often characterized solely in terms of critical thermal maxima (Brattstrom 1963; Claussen 1973; Brown 1975; Bury

*Giant salamanders (*Dicamptodon *spp.) can be found in small western Washington streams/Eric Lund, WDFW*

2008), while stream temperatures in which these species are found are considerably lower than those maxima (invariably ≤61°F [16°C]; e.g., Brattstrom 1963; Welsh 1990; Welsh and Lind 1996; Pollett et al. 2010). Moreover, water temperatures at which these species first become stressed are entirely unknown. Several studies demonstrate higher density or relative abundances (Stoddard and Hayes 2005; Pollett et al. 2010) of amphibians at the margin of their temperature maxima although the mechanism responsible for this effect is unclear. Similar to fish, identifying critical effects of temperatures on amphibians is complicated by a host of potential variables, e.g., younger life stages frequently have cooler temperature requirements (de Vlaming and Bury 1970; Brown 1975) than older life stages. Importantly, very little research has been conducted on the effects of altered thermal regimes on the timing of life history events, such as the time of hatching, and indirect effects, such as possible ecological temporal mismatches, altered predator-prey relations, and disease frequency and effect (e.g., Hari et al. 2006), that would be useful for identifying important conservation and management actions.

Amphibians and reptiles that breed or utilize riverine off-channel stillwater habitats (e.g., oxbow lakes and permanent and seasonally flooded riverine wetlands) associated with stream networks generally have warmer thermal requirements than instream breeding amphibians. For stillwater-breeding amphibians, thermal requirements vary as a function of the thermal tolerance limits of a specific life stage (often eggs and embryos), which determine the seasonal interval in which they lay eggs. For example, Northern Red-legged Frog *Rana aurora* embryos have the lowest critical thermal maximum of any North American frog (about 68°F or 20°C) (Licht 1971), and they oviposit in late winter, at which time surface water temperatures of stillwater habitats rarely if ever put its embryos at risk. Except for our two turtle species (Western Pond Turtle *Actinemys marmorata* and Western Painted Turtle *Chryemys picta*) and selected garter snake species *Thamnophis*, especially the Common Garter Snake *T. sirtalis* and the Western Terrestrial Garter Snake *T. elegans,* few reptiles utilize aquatic habitats in the Pacific Northwest. These reptiles have

Riparian corridor/Ned Pittman, WDFW

thermal requirements that are more elevated than the temperatures generally found in stillwater habitats except during the warmest part of summer. For this reason, aerial basking (basking out of water) at daytime temperatures that are warmer than water temperature can be critical for these species for effective digestion of food or production of eggs. This pattern is especially important for turtles that need aquatic aerial basking sites isolated from predators, which in most stillwater habitats are provided by large downed wood with relatively level accessible surfaces above the water line (Holland 1994).

4.3.3. Invertebrates

Few studies have examined the effects of altered thermal regimes on stream-associated invertebrates in the Pacific Northwest. However, broad relationships between the geographic distributions of invertebrates and their thermal tolerance have been reported (Beschta et al. 1987; Moulton et al. 1993) as have changes in emergence timing and asynchronous emergence among sexes due to small increases in summer temperatures (Li et al. 2011), demonstrating the importance of temperature to invertebrates. Stream temperature can affect invertebrate density, growth, size at maturation, and sex ratios and the effects often differ among species (Hogg and Williams 1996). Altered stream thermal regimes appear to have important effects on many invertebrates and, subsequently, on stream food webs. Indirect effects such as altered predator-prey relations and disease frequency may also be important but have not been well studied. Riparian ecosystem management for aquatic invertebrates is also constrained by a general lack of information on their geographic distribution.

4.3.4. Species Sensitivity Summary

Management of vegetation in riparian ecosystems can affect stream temperatures and, subsequently, fish, amphibian, and invertebrate abundance and survival. Stream temperature influences important food web and energetic interactions between fish, amphibians, and invertebrates in multiple direct ways and likely in more subtle but relatively unstudied indirect ways (Baxter et al. 2005). The effects of thermal regime has received little study for amphibians and invertebrates. Fortunately, our understanding of the mechanisms that affect stream thermal regime can provide useful guidance in the absence of better information.

4.4. Land Use Effects

4.4.1. Urbanization

There have been very few studies specifically relating urbanization to riparian ecosystem function and subsequent effects on stream thermal regimes (Paul and Meyer 2001). Clearly, urbanization related changes to riparian areas can be similar to riparian changes associated with agriculture and forestry as far as stream temperature is concerned, e.g., removal of shade, reduction in large wood inputs. Urban areas have additional issues that can affect stream temperatures including heat island effects, and inputs of wastewater and runoff from impervious surfaces (Kinouchi et al. 2006; Hester and Doyle 2011). Some of these complicating factors may contribute to the absence of detectable effects on stream temperatures studies in some comparative studies (Wahl et al. 2013). Nonetheless, urbanization is considered to have adverse effects on stream temperatures (LeBlanc et al. 1997; Finkenbine 1998), especially when riparian functions are affected (Booth et al. 2001). To protect stream thermal regimes in urban settings, conservation efforts should focus on maintaining riparian functions, mitigating anthropogenic changes to hydrology, and preserving watershed connectivity (e.g., movement of water, sediment, wood, nutrients, and species).

4.4.2. Agriculture

In comparison to urbanization studies, studies of the effects of agricultural practices on riparian ecosystems or subsequent effects on stream thermal regimes are fairly abundant, especially for grazing lands (e.g., Li et al. 1994; Tait et al. 1994; Maloney et al. 1999; Zoellick 2004) and crop lands (e.g., Waite and Carpenter 2000). Studies examining the effects of agriculture, particularly livestock grazing, generally find stream temperature increases associated with disturbance of the riparian ecosystem (Zoellick 2004; Webb et al. 2008). Livestock use of streamside areas can increase a stream's exposure to direct sunlight by reducing shade from streamside vegetation (Li et al. 1994). Livestock can also reduce the amount of undercut banks and increase stream channel width through trampling and vegetation removal (Belsky et al. 1999; Chapter 2). Channel widening can also result in reduced depth, which can exacerbate warming (Poole and Berman 2001). However, results of observational studies are mixed, with occasional absences of predicted detectable increases in stream temperature due to changes in riparian ecosystems as a result of agriculture (e.g., Tait et al. 1994) and wide variation in effect on riparian vegetation among locations and types of agriculture (see Maloney et al. 1999). Importantly, Zoellick (2004) found that riparian vegetation can recover after removal of the disturbance agent (e.g., livestock). For crop lands, the importance of riparian ecosystem attributes, e.g., vegetation width, height and density, on stream temperatures have been documented (Waite and Carpenter 2000) and, similar to grazed lands, effects differ widely among locations (Benedict and Shaw 2012).

> *Studies of the effects of land management on riparian functions and stream temperatures confirm the importance of riparian ecosystems to stream thermal regimes.*

4.4.3. Forestry

Relative to urbanization and agriculture, many studies in the Pacific Northwest have examined the effects of forest management on stream temperatures. Decades of research have frequently documented increased stream temperature due to forestry practices (Olson and Rugger 2007), and our understanding of the mechanisms by which forest practices affect stream temperatures is well developed (Beschta et al. 1987; Groom et al. 2011). Most studies find that forestry, especially when riparian vegetation is disturbed, results in higher stream temperatures than in less disturbed locations (Li et al. 1994), however, exceptions to this pattern are not uncommon. For example, Liquori and Jackson (2001) found stream temperatures 3 to 7°F (1.5 to 4°C) greater in stream reaches surrounded by dense riparian forests compared to downstream reaches adjacent to scrub-shrub riparian communities.

Many of the studies of the effects of forestry on stream temperatures include consideration of some of the components and structures that affect stream thermal regimes identified by Boyd and Kasper (2003) and Poole and Berman (2001). For example, Liquori and Jackson (2001) suggest that their results are due to lower width-to-depth ratio, higher rates of hyporheic or groundwater exchange, and greater amounts of shade in shrub-scrub associated reaches. Maloney et al. (1999) found that much of the variation in their results (in mostly agricultural settings) was explained by reach length, shade, elevation, aspect, and especially by stream gradient and air temperature. Similarly, Kiffney et al. (2003) and Janisch et al. (2012) described the influence of groundwater and hyporheic exchange on their results.

4.4.4. Land Use Summary

Studies of the effects of land management, especially the many studies of forestry, on riparian functions and stream temperatures confirm the importance of riparian ecosystems to stream thermal regimes. Additionally, they emphasize the high degree of variability among locations of the functions that affect stream thermal regimes, even among adjacent stream reaches, that so often produce variable results. Unfortunately, studies that examined the effects of land management on the complete thermal regime of a stream are lacking. Such studies could be especially valuable because the cumulative effects of riparian ecosystem or upland management on stream temperatures can dominate local effects (Chapter 8). Further, most studies use only a few statistics that describe peak or average temperatures, usually during summer. Such statistics are useful, but have limited ability to address dynamic, water quality parameters including temperature (Poole et al. 2004) related to species needs. Thus, predicting the effects of land management on other thermal attributes (e.g., duration or frequency of stressful temperatures) or during other seasons when thermal exposure can significantly affect some species (e.g., overwinter incubation) is difficult. Similarly, even cumulatively, the relatively small number of studies and their skewed geographic distribution (i.e., most are on low-order streams in maritime regions) constrain our ability to provide broad and precise generalizations or make accurate predictions to other locations. Conservation efforts for maintaining stream thermal regimes must be based on our understanding of direct and indirect mechanisms of stream heating affected by human activities, wise use of statistical models relating shade and stream temperature across the watershed until mechanistic models become available, and preserving watershed connectivity (e.g., movement of water, sediment, wood, nutrients, and species).

Light through alder tree canopy/Ned Pittman, WDFW

4.5. Conclusions

Scientific understanding of the structure and function of streams and the importance of thermal regimes has improved substantially since the publication of Knutson and Naef (1997). This understanding underscores the need to consider temperature statistics other than the daily maximum temperature during summer. We now have mechanistic, deterministic models that can accurately predict changes in stream temperatures for specific locations due to specific management actions in the riparian ecosystem. We are also better able to set directional or qualitative expectations for cumulative effects on temperatures at the spatial extent of watersheds. However, quantitative information needed to run mechanistic, deterministic models to make site-scale predictions (e.g., surface water and groundwater flows and temperatures) is expensive to collect and thus are rarely used in management settings. Further, even the most capable stream temperature predictive models are limited by the assumption that the channel structure remains relatively stable (e.g., cross-section shape will not change) even though there is strong evidence that riparian and some types of upslope management affects channel stability (Chapters 2 and 3) at least in some reach types.


Giant salamander (Dicamptodon *spp.*) *egg cluster/Jack Armstrong, WDFW*

Scientific models and most observational studies find that shade from riparian vegetation can affect stream thermal regimes, particularly by lowering peak temperatures during summer. Recent monitoring studies have found substantial variation in the thermal sensitivity of apparently similar stream reaches. The use of improved monitoring tools (e.g., digital temperature loggers), more robust study designs e.g., Before-After-Control-Impact designs (Eberhardt 1976; Stewart-Oaten et al. 1986), more powerful statistical analyses (Gomi et al. 2006; Isaak et al. 2014), and longer study durations have improved our ability to detect effects of riparian management actions on stream temperatures. Importantly, better understanding of the structure and function of streams, especially the importance of surface, ground, and hyporheic water flows on stream temperatures, has allowed scientists to identify likely causes for failures to detect effects of riparian shade reductions on stream temperatures.

Scientific understanding of the importance of stream temperatures on aquatic species has improved substantially over the past two decades, especially for salmonids. Most studies provide useful descriptions of some direct effects of exposure to high temperatures on specific life stages. Laboratory and field studies demonstrated the importance of temperature on fish survival and allowed for comparisons among species, populations, and life stages, and thus the identification of those that are likely most sensitive to heat stress. They have also demonstrated substantial differences in sensitivity among species, populations, and life stages which make developing simple thermal standards challenging and likely inadequate for management. Several recent studies have described effects of prolonged exposure to elevated, but non-lethal, temperatures and altered thermal regimes. These studies demonstrate some of the complexity of this management challenge. Even when altered thermal regimes do not have immediate lethal effects on species, there are costs or trade-offs, such as changes in growth, survival, productivity, and energetic requirements. Detecting such effects is beyond the scope of most studies and will often require long-term, comprehensive monitoring of populations through several life cycles. Such studies will likely need to be conducted at spatial

Salmon eggs in Morse Creek/Steve Boessow, WDFW

extents and durations that account for fish behavior, such as daily and seasonal movement among locations where temperature regimes differ.

Relatively few studies that describe the effects of temperatures or thermal regimes on reptiles, amphibians or invertebrates have been completed. Most studies of these taxonomic groups have simply identified seasonal thermal preferences in the field or the effects of acute temperature in laboratory settings. Such studies are valuable for identifying thermally sensitive species, but are insufficient for management. Further, we found no study that addressed ecological interactions that might be mediated by riparian condition on a stream's thermal regime.

Studies generally show that the effects of urbanization, agriculture, and forestry on watershed and riparian ecosystems result in warmer summer stream temperatures, an expectation consistent with the conceptual model. Importantly, these land uses often include surface water routing to streams (e.g., roads and ditches) that can change the timing, intensity, duration, and temperatures of stream flows. The changes in temperature detected differed among regions, land uses, and studies. The relatively low number of studies and high variability among study results limits our ability to accurately predict outcomes in unstudied locations and precludes us from reliably calculating a mean (i.e., expected) resulting temperature change, even for a specific temperature statistic. Of the studies we reviewed, most were designed to detect an effect using a specific stream temperature statistic (e.g., mean daily maximum summer temperature) rather than changes in the stream's thermal regime. This limits the inferences that can be made, especially from studies in which anticipated temperature changes were not detected, and to make reliable, precise generalizations from study results.

Although we have sound understanding of the composition, structure, and functions that affect stream thermal regimes, substantial uncertainty remains regarding the effects of any specific riparian management action on stream thermal regimes in most locations. That uncertainty is amplified by our relative lack of understanding at large spatial and long temporal extents for which few studies have been completed. Recovery of stream temperatures due to shade from vegetation regrowth can occur in less than a decade following disturbance (D'Souza et al. 2011), but recovery rates differ widely among locations due to differences in stream size and components that control rates of vegetation re-establishment and growth (Gecy and Wilson 1990; D'Souza et al. 2011). Although scientists are very certain that temperatures and thermal regimes are important to many aquatic species, relatively little is known about ecological interactions (e.g., predator-prey relations), behaviors (e.g., movement among locations of different temperature), and indirect effects (e.g., effects on the rates and severity of disease). These interactions may be very important. For example, Lawrence et al. (2014) suggest that riparian restoration in some locations might prevent extirpation of Chinook Salmon and restrict the expansion of non-native fish in some locations by maintaining relatively cool stream temperatures. Such uncertainties often compound and should be addressed explicitly when making management decisions, especially when the decisions can have important, lasting effects (Harwood and Stokes 2003). Management of riparian ecosystem composition (e.g., vegetation species, side channels) and structure (e.g., longitudinal and lateral connectivity) can importantly benefit aquatic species. Management for such beneficial effects might prove especially important as our climate continues to change.

The state of the science regarding stream thermal regimes is uneven. On the one hand, we have a very good understanding of the physics of stream heating and cooling. We understand the relative magnitudes of direct and indirect effects of ecosystem changes on stream thermal regimes and have the ability to quantify these effects given enough time and resources. Shade provided by riparian vegetation is generally considered the single most important, albeit not the only, process of the system that humans can directly effect. These findings are undisputed. On the other hand, because thermal regimes are affected by a variety of factors and their interactions, stream temperatures are difficult to precisely predict on the basis on any single factor alone (e.g., shade). Thus, predicting stream temperature based solely on shade removal or buffer width will likely always suffer from imprecision. In addition to this prediction challenge, and despite the fact that we know suitable water temperatures are critical to the persistence of aquatic species, we are only now beginning to understand how aquatic organisms interact with and are impacted by changes in the stream thermal regime. The combination of what we know and do not know about how changes to stream thermal regimes affect aquatic species and how human activities can affect those regimes makes conservation in the face of a changing climate particularly challenging.

4.6. Literature Cited[1]

Abbe, T.B., and D.R. Montgomery. 1996. Large woody debris jams, channel hydraulics and habitat formation in large rivers. Regulated Rivers Research & Management 12:201-221. (i)

Allan, J.D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology and Systematics 35:257-284. (i)

Allen, M., and L. Dent. 2001. Shade conditions over forested streams in the Blue Mountain and Coast Range georegions of Oregon. ODF Technical Report #13. Oregon Department of Forestry.

Allen, D., W.E. Dietrich, P.F. Baker, F. Ligon, and B. Orr. 2007. Development of a mechanistically-based, basin-scale stream temperature model: applications to cumulative effects modeling. Pages 11-24 *in* R.B. Standiford, G.A. Giusti, Y. Valachovic, W.J. Zielinski, M.J. Furniss, technical editors. Proceedings of the Redwood Region forest science symposium: what does the future hold? PSW-GTR-194. U.S. Forest Service, Pacific Southwest Research Station, Albany, California. (viii)

Alvarez, S.M., J.K. Johnson, and R. Spafford. 2004. Reclamation project allows new phase for Moon

Creek. Land and Water 48:10-15. (i)

Anderson, P.D., D.J. Larson, and S.S. Chan. 2007. Riparian buffer and density management influences on microclimate of young headwater forests of western Oregon. Forest Science 53(2): 254-269.

Arismendi, I., S.L. Johnson, J.B. Dunham, R. Haggerty, and D. Hockman-Wert. 2012. The paradox of cooling streams in a warming world: regional climate trends do not parallel variable local trends in stream temperature in the Pacific Continental United States. Geophysical Research Letters 39:L10401. (i)

Arrigoni, A.S., G.C. Poole, L.A.K. Mertes, S.J. O'Daniel, W.W. Woessner, and S.A. Thomas. 2008. Buffered, lagged, or cooled? Disentangling hyporheic influences on temperature cycles in stream channels. Water Resources Research 44:W09418.

Ayllón D., G.G. Nicola, B. Elvira, I. Parra, and A. Almodóvar. 2013. Thermal carrying capacity for a thermally-sensitive species at the warmest edge of its range. PLoS One 8:e81354. (i)

Baxter, C.V., K.D. Fausch, and W.C. Saunders. 2005. Tangled webs: reciprocal flows of invertebrate prey link streams and riparian zones. Freshwater biology 50:201-220. (i)

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419–431. (i)

Benedict, C., and J. Shaw. 2012. Agricultural waterway buffer study. Washington State University Extension, Bellingham, Washington. (viii)

Benyahya L., D. Caissie, N. El-Jabi, and M.G. Satish. 2010. Comparison of microclimate vs. remote meteorological data and results applied to a water temperature model (Miramichi River, Canada). Journal of Hydrology 380:247–259. (i)

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. Pages 191-232 *in* E.O. Salo and T.W. Cundy, editors. Streamside management: forestry and fishery interactions. Contribution No. 57. Institute of Forest Resources, University of Washington, Seattle. (viii)

Bilby, R.E., and P.A. Bisson. 1987. Emigration and production of hatchery Coho Salmon (*Oncorhynchus kisutch*) stocked in streams draining an old-growth and clear-cut watershed. Canadian Journal of Fisheries and Aquatic Sciences 44:1397-1407. (i)

Bisson, P.A., J.M. Buffington, and D.R. Montgomery. 2006. Valley segments, stream reaches, and channel units. Pages 23-49 *in* F.R. Hauer and G.A. Lamberti, editors. Methods in stream ecology. Elsevier, Burlington, Massachusetts. (i)

Bisson, P.A., S.M. Claeson, S.M. Wondzell, A.D. Foster, and A. Steel. 2013. Evaluating headwater stream buffers: lessons learned from watershed-scale experiments in southwest Washington. Pages 165-184 *in* P.D. Anderson and K.L. Ronnenberg, editors. Density management in the 21st Century: west side story. PNW-GTR-880. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Bobba, A.G., P.A. Chambers, and J. Spoelstra. 2010. Environmental impacts in arid and semiarid floodplains. Journal of Environmental Hydrology 18:1-15. (i)

Booth, D.B., J.R. Karr, S. Schauman, C.P. Konrad, S.A. Morley, M.G. Larson, P.C. Henshaw, E.J. Nelson, and S.J. Burges. 2001. Urban stream rehabilitation in the Pacific Northwest. Final report of EPA Grant Number R82-5284-010. University of Washington, Seattle (viii)

Booth, D.B., K.A. Kraseski, and C.R. Jackson. 2014. Local-scale and watershed-scale determinants of summertime urban stream temperatures. Hydrological Processes 28:2427-2438. (i)

Bowler, D.E., R. Mant, H. Orr, D.M. Hannah, and A.S. Pullin. 2012. What are the effects of wooded riparian zones on stream temperature? Environmental Evidence 2012:3-9. (i)

Boyd, M. 1996. Heat source: stream temperature prediction. Master's thesis. Oregon State University, Corvallis, Oregon. (i)

Boyd, M., and B. Kasper. 2003. Analytical methods for dynamic open channel heat and mass transfer:

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[1] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

methodology for heat source model Version 7.0. Available: **https://www.oregon.gov/deq/FilterDocs/heatsourcemanual.pdf.** (January 2019). (viii)

Brannon, E.L., M.S. Powell, T.P. Quinn, and A. Talbot 2004. Population structure of Columbia River basin Chinook Salmon and steelhead trout. Reviews in Fisheries Science 12:99–232. (i)

Brattstrom, B.H. 1963. A preliminary review of the thermal requirements of amphibians. Ecology 44:238-255. (i)

Brett, J.R. 1952. Temperature tolerance in young Pacific salmon, genus *Oncorhynchus*. Journal of the Fisheries Research Board of Canada 9:265-323. (i)

Brett, J.R. 1971. Energetic responses of salmon to temperature. A study of some thermal relations in the physiology and freshwater ecology of Sockeye Salmon (*Oncorhynchus nerka*). American Zoologist 11:99–113. (i)

Brett, J.R., J.E. Shelbourn, and C.T. Shoop. 1969. Growth rate and body composition of fingerling Sockeye Salmon, *Oncorhynchus nerka,* in relation to temperature and ration size. Journal of the Fisheries Research Board of Canada 26:2363–2394. (i)

Brown, H.A. 1975. Temperature and the development of the Tailed Frog, *Ascaphus truei.* Comparative Biochemistry and Physiology 50A(2):397-405. (i)

Buffington, J.M., D.R. Montgomery, and H.M. Greenberg. 2004. Basin-scale availability of salmonid spawning gravel as influenced by channel type and hydraulic roughness in mountain catchments. Canadian Journal of Fisheries and Aquatic Sciences 61:2085-2096. (i)

Burkholder, B.K., G.E. Grant, R. Haggerty, T. Khangaonkar, and P.J. Wampler. 2008. Influence of hyporheic flow and geomorphology on temperature of a large, gravel-bed river, Clackamas River, Oregon, USA. Hydrological Processes 22:941-953. (i)

Bury, R.B. 2008. Low thermal tolerances of stream amphibians in the Pacific Northwest: implications for riparian and forest management. Applied Herpetology 5:63-74. (i)

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51:1389–1406. (i)

Caissie, D. 2016. River evaporation, condensation and heat fluxes within a first order tributary of Catamaran Brook (New Brunswick, Canada). Hydrological Processes 30:1872–1883. (i)

Claussen, D.L. 1973. The thermal relations of the Tailed Frog, *Ascaphus truei,* and the Pacific Treefrog, *Hyla regilla*. Comparative Biochemistry and Physiology 44A(1):137-153. (i)

Coffin, C., S. Lee, and D. Garland. 2011. Bear-Evans Watershed temperature, dissolved oxygen and fecal coliform bacteria total maximum daily load: water quality implementation plan. Publication No.11-10-024. Washington Department of Ecology, Bellevue, Washington. (viii)

Cole, E., and M. Newton. 2013. Influence of streamside buffers on stream temperature response following clear-cut harvesting in western Oregon. Canadian Journal of Forest Research 43:993-1005. (i)

Cristea, N.C., and S.J. Burges. 2010. An assessment of the current and future thermal regimes of three streams located in the Wenatchee River basin, Washington State: some implications for regional river basin systems. Climatic Change 2010:493-520. (i)

Cristea, N., and J. Janisch. 2007. Modeling the effects of riparian buffer width on effective shade and stream temperature. Washington Department of Ecology, Olympia. (viii)

Crossin, G.T., S.G. Hinch, S.J. Cooke, D.W. Welch, D.A. Patterson, S.R.M. Jones, A.G. Lotto, R.A. Leggatt, M.T. Mathes, J.M. Shrimpton, G. Van Der Kraak, and A.P. Farrell. 2008. Exposure to high temperatures influences the behavior, physiology, and survival of Sockeye Salmon during spawning migration. Canadian Journal of Zoology 86:127-140. (i)

Czarnomski, N., C. Hale, W.T. Frueh, M. Allen, and J. Groom. 2013. Effectiveness of riparian buffers at protecting stream temperature and shade in Pacific Northwest forests: a systematic review. Final Report to the Forest Practices Monitoring Program. Oregon

Department of Forestry, Salem. (viii)

de Vlaming, V.L., and R.B. Bury. 1970. Thermal selection in tadpoles of the Tailed-Frog, *Ascaphus truei.* Journal of Herpetology 4:179-189. (i)

DeWalle, D.R. 2010. Modeling stream shade: riparian buffer height and density as important as buffer width. Journal of the American Water Resources Association 46:323-333. (i)

D'Souza, L.E., M. Reiter, L.J. Six, and R.E. Bilby. 2011. Response of vegetation, shade and stream temperature to debris torrents in two western Oregon watersheds. Forest Ecology and Management 261:2157-2167. (i)

Dugdale, S.J., N.E. Bergeron, and A. St-Hilaire. 2013. Temporal variability of thermal refuges and water temperature patterns in an Atlantic Salmon river. Remote Sensing of Environment 136:358-373. (i)

Eberhardt, L.L. 1976. Quantitative ecology and impact assessment. Journal of Environmental Management 4:27–70. (i)

Ebersole, J.L., W.J. Liss, and C.A. Frissell. 2003a. Thermal heterogeneity, stream channel morphology, and salmonid abundance in northeastern Oregon streams. Canadian Journal of Fisheries and Aquatic Sciences 60:1266-1280. (i)

Ebersole, J.L., W.J. Liss, and C.A. Frissell. 2003b. Cold water patches in warm streams: physicochemical characteristics and the influence of shading. Journal of the American Water Resources Association 39:355-368. (i)

Eliason, E.J., T.D. Clark, M.J. Hague, L.M. Hanson, Z.S. Gallagher, K.M. Jeffries, M.K. Gale, D.A. Patterson, S.G. Hinch, and A.P. Farrell. 2011. Differences in thermal tolerance among Sockeye Salmon populations. Science 332:109-112. (i)

Elmore, W., and J.B. Kauffman. 1994. Riparian and watershed systems: degradation and restoration. Pages 211-232 *in* M. Vavra, W.A. Laycock, and R.D. Piper, editors. Ecological implications of livestock herbivory in the West. Society of Range Management, Denver, Colorado. (viii)

Farrell, A.P., S.G. Hinch, S.J. Cooke, D.A. Patterson, G.T. Crossin, M. Lapointe, and M.T. Mathes. 2008. Pacific salmon in hot water: applying aerobic scope models and biotelemetry to predict the success of spawning migrations. Physiological and Biochemical Zoology 81:697-709. (i)

Fausch, K.D., C.E. Torgersen, C.V. Baxter, and H.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52:1-16. (i)

Finkenbine, J.K., J.W. Atwater, and D.S. Mavinic. 2000. Stream health after urbanization. Journal of the American Water Resources Association 36:1149-1160. (i)

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199-214. (i)

Fullerton, A.H., C. Torgersen, J.J. Lawler, R.N. Faux, E.A. Steel, T.J. Beechie, J.L. Ebersole, and S.J. Leibowitz. 2015. Rethinking the longitudinal stream temperature paradigm—region-wide comparison of thermal infrared imagery reveals unexpected complexity of river temperatures. Hydrological Processes 29:4719-4737. (i)

Garner, G., I.A. Malcolm, J.P. Sadler, and D.M. Hannah. 2014. What causes cooling water temperature gradients in a forested stream reach? Hydrology and Earth System Sciences 18:5361-5376. (i)

Gecy, J.L., and M.V. Wilson. 1990. Initial establishment of riparian vegetation after disturbance by debris flows in Oregon. American Midland Naturalist 123:282-291. (i)

Gomi, T., R.D. Moore, and A.S. Dhakal. 2006. Headwater stream temperature response to clear-cut harvesting with different riparian treatments, coastal British Columbia, Canada. Water Resources Research

42: W08437.

Gomi, T., R.C. Sidle, and D.N. Swanston. 2004. Hydrogeomorphic linkages of sediment transport in headwater streams, Maybeso Experimental Forest, southeast Alaska. Hydrological Processes 18:667-667. (i)

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience 41:540-551. (i)

Groom, J.D., L. Dent, and J. Madsen. 2011. Stream temperature change detection for state and private forests in the Oregon Coast Range. Water Resources Research 47:1-12. (i)

Groot, C., and L. Margolis. 2010. Pacific salmon life histories. University of Washington Press, Seattle. (i)

Hari, R.E., D.M. Livingston, R. Siber, P. Burkhardt-Holm, and H. Guttinger. 2006. Consequences of climate change for water temperature and brown trout populations in alpine rivers and streams. Global Change Biology 12:10-26. (i)

Harwood, J., and K. Stokes. 2003. Coping with uncertainty in ecological advice: lessons from fisheries. Trends in Ecology and Evolution 18:617–622. (i)

Hawkins, C.P., M.L. Murphy, N.H. Anderson, and M.A. Wilzbach. 1983. Density of fish and salamanders in relation to riparian canopy and physical habitat in streams of the northwestern United States. Canadian Journal of Fisheries and Aquatic Sciences 40:1173-1185. (i)

Hendry, A.P., J.E. Hensleigh, and R.R. Reisenbichler. 1998. Incubation temperature, developmental biology, and the divergence of Sockeye Salmon (*Oncorhynchus nerka*) within Lake Washington. Canadian Journal of Fisheries and Aquatic Sciences 55:1387–1394. (i)

Hester, E.T., and M.W. Doyle. 2011. Human impacts to river temperature and their effects on biological processes: a quantitative synthesis. Journal of the American Water Resources Association 47:571–587. (i)

Hinch, S.G., S.J. Cooke, A.P. Farrell, K.M. Miller, M. LaPointe, and D.A. Patterson. 2012. Dead fish swimming: a review of research on the early migration and high premature mortality in adult Fraser River Sockeye Salmon (*Oncorhynchus nerka*). Journal of Fish Biology 81:576-599. (i)

Hodgson, S., and T.P. Quinn. 2002. The timing of adult Sockeye Salmon migration into fresh water: adaptations by populations to prevailing thermal regimes. Canadian Journal of Zoology 80:542-555. (i)

Hogg, I.D., and D.D. Williams. 1996. Response of stream invertebrates to a global-warming thermal regime: an ecosystem-level manipulation. Ecology 77:395-407. (i)

Holland, D.C. 1994. The Western Pond Turtle: habitat and history. Unpublished final report, U. S. Dept. of Energy, Portland, Oregon. (viii)

Holsinger, L., R.E. Keane, D.J. Isaak, L. Eby, and M.K. Young. 2014. Relative effects of climate change and wildfires on stream temperatures: a simulation modeling approach in a Rocky Mountain watershed. Climate Change 124:191-206. (i)

Isaak, D.J., and W.A. Hubert. 2001. A hypothesis about factors that affect maximum summer stream temperatures across montane landscapes. Journal of the American Water Resources Association 37:351-366. (i)

Isaak, D.J., and W.A. Hubert. 2004. Nonlinear response of trout abundance to summer stream temperatures across a thermally diverse montane landscape. Transactions of the American Fisheries Society 133:1254-1259. (i)

Isaak, D.J., C.H. Luce, B.B. Rieman, D.E. Nagel, E.E. Peterson, D.L. Horan, S. Parkes, and G.L. Chandler. 2010. Effects of climate change and wildfire on stream temperatures and salmonid thermal habitat in a mountain river network. Ecological

Applications 20:1350-1371. (i)

Isaak, D.J., E.E. Peterson, J.M. Ver Hoef, S.J. Wenger, J.A. Falke, C. Torgersen, C. Sowder, E.A. Steel, M. Fortin, C.E. Jordan, A.S. Ruesch, N. Som, and P. Monestiez. 2014. Applications of spatial statistical network models to stream data. Wiley Interdisciplinary Reviews—Water 1:277-294. (i)

Isaak, D., S. Wenger, E. Peterson, J. Ver Hoef, D. Nagel, C. Luce, S. Hostetler, J. Dunham, B. Roper, S. Wollrab, G. Chandler, D. Horan, and S. Parkes-Payne. 2017a. The NorWeST summer stream temperature model and scenarios for the western U.S.: a crowd-sourced database and new geospatial tools foster a user community and predict broad climate warming of rivers and streams. Water Resources Research 53:9181-9205. (i)

Isaak, D.J., S.J. Wenger, and M.K. Young. 2017b. Big biology meets microclimatology: defining thermal niches of ectotherms at landscape scales for conservation planning. Ecological Applications 27:977–990. (i)

Isaak, D.J., S. Wollrab, D. Horan, and G. Chandler. 2012. Climate change effects on stream and river temperatures across the Northwest U.S. from 1980–2009 and implications for salmonid fishes. Climatic Change 113:499-524. (i)

Isaak, D.J., M.K. Young, D. Nagel, D. Horan, and M. Groce. 2015. The coldwater climate shield: delineating refugia to preserve salmonid fishes through the 21st Century. Global Change Biology 21:2540-2553. (i)

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber harvest impacts on small headwater stream channels in the coast ranges of Washington. Journal of the American Water Resources Association 37:1533–1549.

Janisch, J.E., S.M. Wondzell, and W. J. Ehinger. 2012. Headwater stream temperature: interpreting response after logging, with and without riparian buffers, Washington, USA. Forest Ecology and Management 270:302-313. (i)

Johnson, F.A. 1971. Stream temperatures in an alpine area. Journal of Hydrology 14:322-336. (i)

Johnson, S.L. 2004. Factors influencing stream temperatures in small streams: substrate effects and a shading experiment. Canadian Journal of Fisheries and Aquatic Sciences 61:913-923. (i)

Kappenman, K.M., M.A.H. Webb, and M. Greenwood. 2013. The effect of temperature on embryo survival and development in Pallid Sturgeon *Scaphirhynchus albus* (Forbes & Richardson 1905) and Shovelnose Sturgeon *S. platorynchus* (Rafinesque, 1820). Journal of Applied Ichthyology 29:1193-1203. (i)

Keefer, M.L., C.A. Peery, and M.J. Heinrich. 2008. Temperature-mediated en route migration mortality and travel rates of endangered Snake River Sockeye Salmon. Ecology of Freshwater Fish 17:136–145. (i)

Keller, E.A., and F.J. Swanson. 1979. Effects of large organic material on channel form and fluvial processes. Earth Surface Processes 4:361-380. (i)

Kibler, K.M., A. Skaugset, L.M. Ganio, and M.M. Huso. 2013. Effects of contemporary forest harvesting practices on headwater stream temperatures: initial response of the Hinkle Creek catchment, Pacific Northwest, USA. Forest Ecology and Management 310:680-691. (i)

Kiffney, P.M., J.S. Richardson, and J.P. Bull. 2003. Responses of periphyton and insect consumers to experimental manipulation of riparian buffer width along headwater streams. Journal of the American Water Resources Association 40:1060-1076. (i)

Kinouchi, T., H. Yagi, and M. Miyamoto. 2006. Increase in stream temperature related to anthropogenic heat input from urban wastewater. Journal of Hydrology 335:78–88. (i)

Knutson, K.L., and V.L. Naef. 1997. Management recommendations for Washington's priority habitats: riparian. Washington Department of Fish and Wildlife, Olympia, Washington. (iii)

Konrad, C.P., D.B. Booth, and S.J. Burges. 2005. Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns: consequences for channel form and streambed disturbance. Water Resources Research 41:W07009. (i)

Lawrence, D.J., B. Stewart-Koster, J.D. Olden, A.S. Ruesch, C.E. Torgersen, J.J. Lawler, D.P. Butcher, and J.K. Crown. 2014. The interactive effects of climate change, riparian management, and a nonnative predator on stream-rearing salmon. Ecological Applications 24:895-912. (i)

LeBlanc, R.T., R.D. Brown, and J.E. FitzGibbon. 1997. Modeling the effects of land use change on the water temperature in unregulated urban streams. Journal of Environmental Management 49:445–469. (i)

Leinenbach, P., G. McFadden, and C. Torgersen. 2013. Effects of riparian management strategies on stream temperature, Part II. Summary report of the Science Review Team submitted to the Interagency Coordinating Subcommittee. Bureau of Land Management, Portland, Oregon. (viii)

Li, H.W., G.A. Lamberti, T.N. Pearsons, C.K. Tait, and J.C. Buckhouse. 1994. Cumulative effects of riparian disturbances along high desert trout streams of the John Day Basin, Oregon. Transactions of the American Fisheries Society 123:627–640. (i)

Li, J.L., S.L. Johnson, and J.B. Sobota. 2011. Three responses to small changes in stream temperature by autumn-emerging aquatic insects. Journal of the North American Benthological Society 30:474-484. (i)

Licht, L.E. 1971. Breeding habits and embryonic thermal requirements of the frogs, *Rana aurora aurora* and *Rana pretiosa pretiosa*, in the Pacific Northwest. Ecology 52:116–124. (i)

Liquori, M., and C.R. Jackson. 2001. Channel response from shrub dominated riparian communities and associated effects on salmonid habitat. Journal of the American Water Resources Association 37:1639–1651. (i)

Luce, C.H., B.P. Staab, M.G. Kramer, S.J., D.J. Isaak, and C. McConnell. 2014. Sensitivity of summer stream temperatures to climate variability in the Pacific Northwest. Water Resources Research 50:1-16. (i)

Lytle, D.A., and N.L. Poff. 2004. Adaptation to natural flow regimes. Trends in Ecology and Evolution 19:94-100. (i)

Malard, F., K. Tockner, M. Dole-Olivier, and J.V. Ward. 2002. A landscape perspective of surface-subsurface hydrological exchanges in river corridors. Freshwater Biology 47:621-640. (i)

Malcolm, I.A., C. Soulsby, and A.F. Youngson. 2002. Thermal regime in the hyporheic zone of two contrasting salmonid spawning streams: ecological and hydrological implications. Fisheries Management and Ecology 9:1-10. (i)

Maloney, S.B., A.R. Tiedemann, D.A. Higgins, T.M. Quigley, and D.B. Marx. 1999. Influence of stream characteristics and grazing intensity on stream temperatures in eastern Oregon. PNW-GTR-459. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Marsh, P.C. 1985. Effect of incubation temperature on survival of embryos of native Colorado River fishes. The Southwestern Naturalist 30:129-140. (i)

May, C.W. 2003. Stream-riparian ecosystems in the Puget Sound lowland ecoregions: a review of best available science. Watershed Ecology, LLC, Poulsbo, Washington. (i)

McCullough, D.A. 1999. A review and synthesis of effects of alterations to the water temperature regime of freshwater life stages of salmonids, with special reference to Chinook Salmon. U.S. Environmental Protection Agency, Region 10, Seattle, Washington. (viii)

McCullough, D.A., J.M. Bartholow, H.I. Jager, R.L. Beschta, E.F. Cheslak, M.L. Deas, J.L. Ebersole, J.S. Foote, S.L. Johnson, K.R. Marine, M.G. Mesa, J.H. Petersen, Y. Souchon, K.F. Tiffan, and D.A. Wurtsbaugh. 2009. Research in thermal biology: burning questions for coldwater stream fishes. Reviews in Fisheries Science 17:90-115. (i)

McCullough, D., S. Spalding, D. Sturdevant, and M. Hicks. 2001. Issue Paper 5. Summary of technical literature examining the physiological effects of temperature on salmonids. EPA-910-D-01-005. U.S. Environmental Protection Agency, Region 10, Seattle, Washington. (viii)

Micheli, E.R., and J.W. Kirchner. 2002. Effects of wet meadow riparian vegetation on streambank erosion. 2. Measurements of vegetated bank strength and consequences for failure mechanics. Earth Surface Processes and Landforms 27:687-697. (i)

Moore, R.D., D.L. Spittlehouse, and A. Story. 2005. Riparian microclimate and stream temperature response to forest harvesting: a review. Journal of the American Water Resources Association 41:813-834. (i)

Moulton, S.R., T.L. Beitinger, K.W. Stewart, and R.J. Currie. 1993. Upper temperature tolerance of four species of caddisflies (Insecta: Trichoptera). Journal of Freshwater Ecology 8:193–198. (i)

Murphy, M.L., and J.D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams in the Cascade Mountains, Oregon. Canadian Journal of Fisheries and Aquatic Sciences 38:137-145. (i)

Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. Transactions of the American Fisheries Society 110:469-478. (i)

NRC (National Research Council). 2002. Riparian areas: functions and strategies for management. National Academy Press, Washington, D.C. (i)

Nyberg, L., O. Calles, and L. Greenberg. 2008. Impact of short-term regulation on hyporheic water quality in a boreal river. River Research and Applications 24:407-419. (i)

Olden, J.D., and R.J. Naiman. 2010. Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity. Freshwater Biology 55:86-107. (i)

Olson, D.H., and C. Rugger. 2007. Preliminary study of the effects of headwater riparian reserves with upslope thinning on stream habitats and amphibians in western Oregon. Forest Science 53:331-342. (i)

Paul, M.J., and J.L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365. (i)

Pollett, K.L., J.G. MacCracken, and J.A. MacMahon. 2010. Stream buffers ameliorate the effects of timber harvest on amphibians in the Cascade Range of southern Washington, USA. Forest Ecology and Management 260:1083-1087. (i)

Poole, G.C., and C.H. Berman. 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management 27:787-802. (i)

Poole, G.C., J.B. Dunham, D.M. Keenan, S.T. Sauter, D.A. McCullough, C. Mebane, J.C. Lockwood, D.A. Essig, M.P. Hicks, D.J. Sturdevant, E.J. Materna, S.A. Spalding, J. Risley, and M. Deppman. 2004. The case for regime-based water quality standards. BioScience 54:155-161. (i)

Poole, G., J. Risley, and M. Hicks. 2001. Issue paper 3: spatial and temporal patterns of stream temperature (revised). EPA-910-D-01-003. U.S. Environmental Protection Agency, Region 10, Seattle, Washington. (viii)

Post, D.M., E.P. Palkovacs, E.G. Schielke, and S.I. Dodson. 2008. Interspecific variation in a predator affects community structure and cascading trophic interactions. Ecology 98:2019–2032. (i)

Pringle, C.M., and F.J. Triska. 2000. Emergent biological patterns in streams resulting from surface-subsurface water interactions at landscape scales. Pages 167-193 *in* J.B. Jones and P.J. Mulholland, editors. Streams and groundwaters. Elsevier, Burlington, Massachusetts. (i)

Reeves, G.H., D.H. Olson, S.M. Wondzell, P.A. Bisson, S. Gordon, S.A. Miller, J.W. Long, and M.J. Furniss. 2018. Chapter 7: The Aquatic Conservation Strategy of the Northwest Forest Plan—a review of the relevant science after 23 years. Pages 461-642 *in* T.A. Spies, P.A. Stine, R. Gravenmier, J.W. Long, and M.J. Reilly, technical coordinators. Synthesis of science to inform land management within the Northwest Forest Plan area. PNW-GTR-966. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Rezende E.L., L.E. Castañeda, and M. Santos. 2014. Tolerance landscapes in thermal ecology. Functional Ecology 28:799–809. (i)

Richter, A., and S.A. Kolmes. 2005. Maximum temperature limits for Chinook, Coho, and Chum Salmon, and steelhead trout in the Pacific Northwest. Reviews in Fisheries Science 13:23–49. (i)

Rykken, J.J., S.C. Chan, and A.R. Moldenke. 2007. Headwater riparian microclimate patterns under alternative forest management treatments. Forest Science 53:270-280. (i)

Sather, N.K., and C.W. May. 2007. Riparian buffer zones in the interior Columbia River Basin: a review of best available science. Battelle Memorial Institute, Marine Sciences Laboratory, Sequim, Washington. (viii)

Sauter, S.T., L.I. Cershaw, and A.G. Maule. 2001. Behavioral thermoregulation by juvenile spring and fall Chinook Salmon, *Oncorhynchus tshawytscha*, during smoltification. Environmental Biology of Fishes 61:295–304. (i)

Sauter, S.T., and P.J. Connolly. 2010. Growth, condition factor, and bioenergetics modeling link warmer stream temperatures below a small dam to reduced performance of juvenile steelhead. Northwest Science 84:369-377. (i)

Schuett-Hames, D., A. Roorbach, and R. Conrad. 2012. Results of the westside type N buffer characterization, integrity and function study final report. Cooperative Monitoring Evaluation and Research Report, CMER 12-1201. Washington Department of Natural Resources, Olympia.

Sinokrot, B.A., and H.G. Stefan. 1993. Stream temperature dynamics: measurements and modeling. Water Resources Research 29:2299-2312. (i)

Sridhar,V., A.L. Sansone, J. Lamarche, T. Dubin, and D.P. Lettenmaier. 2004. Prediction of stream temperature in forested watersheds. Journal of the American Water Resources Association 40:197-213. (i)

Stanford, J.A. 1998. Rivers in the landscape: introduction to the special issues on riparian and groundwater ecology. Freshwater Biology 40:402-406. (i)

Steel, E.A., A. Tillotson, D.A. Larsen, A. Fullerton, K.P. Denton, and B.R. Beckman. 2012. Beyond the mean: the role of variability in predicting ecological effects of stream temperatures on salmon. Ecosphere 3:104 (i)

Stewart-Oaten, A., W.W. Murdoch, and K.R. Parker. 1986. Environmental impact assessment: pseudoreplication in time? Ecology 67:929–940. (i)

Stoddard, M.A., and J.P. Hayes. 2005. The influence of forest management on headwater stream amphibians at multiple spatial scales. Ecological Applications 15:811-823. (i)

Story, A., R.D. Moore, and J.S. Macdonald. 2003. Stream temperatures in two shaded reaches below cutblocks and logging roads: downstream cooling linked to subsurface hydrology. Canadian Journal of Forest Research 33:1383-1396. (i)

Strange, J.S. 2010. Upper thermal limits to migration in adult Chinook Salmon: evidence from the Klamath River basin. Transactions of the American Fisheries Society 139:1091-1108. (i)

Sweeney, B.W., and J.D. Newbold. 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: a literature review. Journal of the American Water Resources Association 50:560-584. (i)

Tait, C.K., J.L. Li, G.A. Lamberti, T.N. Pearsons, and H.W. Li. 1994. Relationships between riparian cover and the community structure of high desert streams. Journal of the North American Benthological Society 13:45–56. (i)

Toledo, Z.O., and J.B. Kauffman. 2001. Root biomass in relation to channel morphology of headwater streams. Journal of the American Water Resources Association 37:1653-1663. (i)

Torgersen, C.E., R. Faux, B.A. McIntosh, N. Poage, and D.J. Norton. 2001. Airborne thermal remote sensing for water temperature assessment in rivers and streams. Remote Sensing of Environment 76:386-398. (i)

Torgersen, C.E., D.M. Price, H.W. Li, and B.A. McIntosh. 1999. Multiscale thermal refugia and stream habitat associations of Chinook Salmon in northwestern Oregon. Ecological Applications 9:301-319. (i)

USEPA 2012. Technical memorandum to the Idaho Department of (state) Lands (IDL) Private Lands Forest Practices Act (FPA) Shade Subcommittee. Available: ftp://ftp.epa.gov/ reg10ftp/forestry/. (June 2015). (viii)

Ver Hoef, J.M., E.E. Peterson, and D. Theobald. 2006. Spatial statistical models that use flow and stream distance. Environmental and Ecological Statistics 13:449–464. (i)

Wahl, C.M., A. Neils, and D. Hooper. 2013. Impacts of land use at the catchment scale constrain the habitat benefits of stream riparian buffers. Freshwater Biology 58:2310–2324. (i)

Waite, I.R., and K.D. Carpenter. 2000. Associations among fish assemblage structure and environmental variables in Willamette Basin streams, Oregon. Transactions of the American Fisheries Society 129:754–770. (i)

Ward, J.V., F. Malard, and K. Tockner. 2002. Landscape ecology: a framework for integrating pattern and process in river corridors. Landscape Ecology 17(Supplement 1):35–45. (i)

Ward, J.V., and J.A. Stanford. 1982. Thermal responses in the evolutionary ecology of aquatic insects. Annual Review of Entomology 27:97-117. (i)

Ward, J.V., and J.A. Stanford. 1995. Ecological connectivity in alluvial river ecosystems and its disruption by flow regulation. Regulated Rivers: Research & Management 11:105-119. (i)

Webb, B.W., D.M. Hannah, R.D. Moore, L.E. Brown, and F. Nobilis. 2008. Recent advances in stream and river temperature research. Hydrological Processes 22:902–918. (i)

Weitkamp, L.A., D.J. Teel, M. Liermann, S.A. Hinton, D.M. Van Doornik, and P.J. Bentley. 2015. Stock-specific size and timing at ocean entry of Columbia River juvenile Chinook Salmon and steelhead: implications for early ocean growth. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 7:370-392. (i)

Welsh, H.H. 1990. Relictual amphibians and old-growth forests. Conservation Biology 4:309-319. (i)

Welsh, H.H., Jr., G R. Hodgson, B.C. Harvey, and M.E. Roche. 2001. Distribution of juvenile Coho Salmon in relation to water temperatures in tributaries of the Mattole River, California. North American Journal of Fisheries Management 21:464-470. (i)

Welsh, H.H., Jr., and A.J. Lind. 1996. Habitat correlates of the Southern Torrent Salamander, *Rhyacotriton variegatus* (Caudata: Rhyacotritonidae), in northwestern California. Journal of Herpetology 30:385-398. (i)

Wiens, J.A. 2002. Riverine landscapes: taking landscape ecology into the water. Freshwater Biology 47:501–515. (i)

Winder, M., and D.E. Schindler. 2004. Climate change uncouples trophic interactions in an aquatic ecosystem. Ecology 85:2100–2106. (i)

Wondzell, S.M. 2012. Hyporheic zones in mountain streams: physical processes and ecosystem functions. Stream Notes (Jan. - Apr.):1-10. Stream Systems Technology Center, Rocky Mountain Research Station, Fort Collins, Colorado. (i)

Wondzell, S.M., and M.N. Gooseff. 2013. Geomorphic controls on hyporheic exchange across scales: watersheds to particles. Pages 203-218 *in* J. Shroder, editor-in-chief, and E. Wolh, editor. Treatise on Geomorphology. Academic Press, San Diego, California.

Wright, K.K., C.V. Baxter, and J.L. Li. 2005. Restricted hyporheic exchange in an alluvial river system: implications for theory and management. Journal of the North American Benthological Society 24:447-460. (i)

Zaugg, W.S., and H.H. Wagner. 1973. Gill ATPase activity related to parr-smolt transformation and migration in steelhead trout (*Salmo gairdneri*): influence of photoperiod and temperature. Comparative Biochemistry and Physiology 45:955-965. (i)

Zoellick, B.W. 2004. Density and biomass of redband trout relative to stream shading and temperature in southwestern Idaho. Western North American Naturalist 64:18–26. (i)

Long-toed salamander Ambystoma macrodactylum*/Scott Fitkin, WDFW*

Chapter 5. Pollutant Removal

By: Jeffrey L. Ullman and George F. Wilhere

5.1. Introduction

The capability of riparian areas to remove certain pollutants from runoff has been known to scientists for over 60 years (e.g., Trimble and Sartz 1957). Understanding the complex processes and associated factors that affect the pollutant removal function of riparian areas is essential for developing management strategies that will protect aquatic ecosystems. Management of riparian areas for pollutant removal differs from the other ecological functions discussed in this document in that the primary focus is on mitigating activities occurring outside the riparian area. The pollutant removal function is unique in that it only exists in the presence of human activities that generate polluted water, and it is only necessary when runoff from upland activities threaten to degrade water quality.

This chapter summarizes key information relevant to riparian pollutant removal function. First, a general background is presented on the main processes and related site-level factors affecting the transport and fate of nonpoint source pollutants in riparian areas. Next, the biological impacts, common sources, *in situ* chemistry, transport pathways, and mitigation are reviewed for five types of pollutants—sediments, excess nutrients, metals, pesticides and other organic compounds, and pathogens. Last, a summary of what is currently known about the relationships between riparian buffer width and pollutant removal is provided. Riparian buffer is not synonymous with riparian area. Riparian area is an ecological designation that is synonymous with riparian ecosystem (see Chapter 1). Riparian buffer is a management designation, and a buffer may be narrower or wider than the riparian ecosystem.

Yellowjacket Creek/George Fornes, WDFW

This chapter only addresses nonpoint-source chemical pollutants that interact with riparian areas. Hence, it does not deal with "thermal pollution" effects on water temperature (see Chapter 4), which is another important water quality parameter. Similarly, it does not deal with pollutants routed via roads, ditches, and drains into urban and suburban stormwater runoff systems, or with polluted water routed to rivers and streams via agricultural tile drainage.

While the scientific literature reviewed here includes information collected during the 1970s and 1980s, when scientists first started to rigorously investigate the removal of pollutants by riparian areas (Correll 1997), more recent findings are highlighted. Decades of research have revealed that the ecosystem structures and processes responsible for pollutant removal functions of riparian areas are complex, spatially and temporally variable, and dependent on site-level environmental conditions. This complexity and variability are the main reasons for inconsistent results found in the published research and the lack of definitive recommendations regarding the structure and size of riparian buffers required for pollutant removal. Nevertheless, general trends in research findings can guide policy makers and resource managers to management strategies that address these uncertainties and lead to adequate protection of water quality.

5.2. The Pollutant Removal Function of Riparian Areas

Water quality refers to the physical, chemical, and biological characteristics of water that indicate a waterbody's suitability to meet human needs or habitat requirements for fish and wildlife. Riparian areas exert a significant influence on water quality due to their position between terrestrial and aquatic ecosystems. Pollutants in dissolved, chemical, or suspended solid forms are transported by water from uplands to surface waterbodies, and while passing through riparian areas contaminated water undergoes a variety of physical, chemical, and biological processes that reduce pollutant concentrations. The processes in riparian areas affecting pollutant transport and fate are complex and often interrelated. Riparian areas slow surface runoff and increase infiltration of water into the soil, thereby enhancing both deposition of solids and filtration of water-borne pollutants. Riparian areas also intercept and act on contaminants in subsurface flow through dilution, sorption[1], physical transformation, chemical degradation, or volatilization by various biogeochemical processes and through uptake and assimilation by plants, fungi, and microbes.

There is overwhelming evidence in the scientific literature that riparian buffers reduce nonpoint source water pollution for a variety of pollutants—including sediments, excess nutrients, metals, organic compounds such as pesticides, and pathogens. Management activities can enhance the physical, chemical, and biological mechanisms that reduce the flow of polluted waters to river and streams.

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[1] Sorption or "to sorb" is a physical and chemical process by which one substance becomes attached to another. It encompasses two processes adsorption and absorption. Adsorption is the binding of molecules or particles to a substance's surface. The reverse of adsorption is desorption. Absorption is the filling of minute pores in a substance by molecules or particles.

Riparian buffer widths for pollution removal need not be one-size-fits-all/U.S. Dept. of Agriculture, National Agriculture Imagery Program

However, the science is very clear that results vary considerably depending on local conditions, and that the configuration of a riparian buffer may effectively remove one type of pollutant while having minimal effect on another. Although riparian management can reduce significantly the quantity of pollutants entering surface waters, riparian areas are not sufficient by themselves to fully mitigate pollution from nonpoint sources. Thus, it is critical to view pollution control from a holistic, watershed perspective that also manages contaminated runoff at its source, i.e., upland areas subjected to various intensive land uses: forestry, agriculture, commercial, and urban, suburban, and rural residential. Nevertheless, maintaining functional riparian areas is essential for reducing nonpoint source pollution and must be a key component in any suite of management actions focused on water quality.

5.3. Processes Influencing the Pollutant Removal Function

Riparian areas are consistently shown to remove a variety of nonpoint source pollutants, such as those originating from agricultural lands, urban areas, construction sites, and forestry operations. However, the effectiveness of riparian buffers is highly variable (Hickey and Doran 2004; Lee et al. 2004, Anbumozhi et al. 2005; Zhang et al. 2010). Inconsistencies among research results can be explained by the multi-dimensional heterogeneity of riparian ecosystems; the complex, interconnected processes that act on contaminant transport and fate; and the variety of research methods employed. Many site-specific factors contribute to this variability, including riparian buffer width, slope, topography, surface and subsurface hydrology, soil properties, riparian vegetation, and adjacent upland characteristics. Of all these factors hydrology, soil, and vegetation play dominant roles governing pollutant transport and fate (NRC 2002). The following sections discuss the effects of these three site-level characteristics on the pollutant removal function of riparian areas.

5.3.1. Hydrology

Hydrologic regimes at the terrestrial-aquatic interface impart a particularly significant influence on the attenuation of pollutants by riparian areas. Three principal pathways govern water movement through these systems: 1) overland flow and shallow subsurface flow from adjacent uplands, 2) groundwater flow, and 3) instream water movement that includes hyporheic exchange as well as flood conditions (NRC 2002). Each of these flow paths can act as conduits for pollutant transport to aquatic ecosystems depending on site-specific conditions and pollutant type. Furthermore, seasonal stream flows, fluctuating water tables, and intermittent flooding create a dynamic environment that subjects pollutants to a range of biogeochemical processes.

Upland sources of water often originate as precipitation. During and immediately after a precipitation event, water from upland areas follows different flow paths to the stream channel, each contributing differently to pollutant transport and fate. Overland flow can be either infiltration-excess[2] (Hortonian overland flow) or saturation-excess[3] overland flow. Water that enters the soil profile can also travel downslope in shallow subsurface flow (interflow) through the unsaturated zone[4]. Depending on site-specific conditions, water in one of these flow paths may switch to another. Collectively these flow paths are commonly termed hillslope runoff. Under certain circumstances, such as in the presence of deep, well-drained soils that facilitate percolation, portions of the shallow subsurface flow can reach the groundwater table and resume transport in the groundwater system. These pathways extend into riparian areas before reaching a surface waterbody.

As overland flow travels through the landscape it can pick up pollutants that can be in either particulate or soluble form. When water infiltrates the soil surface, suspended particulate matter and associated contaminants are entrapped at the soil surface, almost exclusively limiting the form of pollutants that are transported through the subsurface to soluble forms that remain in solution. However, recent research highlights the significance of colloid-facilitated transport as an additional means by which contaminants can move through the subsurface (Sen and Khilar 2006; Bradford and Torkzaban 2008). Colloidal forms (particles ranging from 1 to 1,000 nm) can remain dispersed in solution and exhibit characteristics resembling solutions. This behavior can consequently mobilize substances that tend to sorb to soil particles or organic matter that would otherwise not be able to move through the subsurface. While the fundamental physics and chemistry of this mechanism is understood, the contribution of colloids to pollutant migration through the soil profile is unresolved. Essentially no literature exists on the role of colloids in riparian areas, contributing to uncertainty in predicting the ultimate fate of associated pollutants.

The fundamental variables of interest regarding groundwater are flow rate and travel time, which influence natural chemical and biological processes that remove soluble pollutants, such as sorption to soils, microbial degradation, and plant uptake (Vogt et al.

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[2] Infiltration-excess overland flow occurs when water enters a soil system faster than the soil can absorb or transport it, such as when precipitation exceeds the infiltration capacity of the soil.

[3] Saturation-excess overland flow occurs when the soil becomes saturated, and any additional precipitation results in runoff.

[4] The unsaturated zone is also known as the vadose zone. It extends from the ground surface to the water table. Beneath the water table is the zone of saturation.

2012). As groundwater approaches the stream margin biogeochemical activity increases, and this gradient contributes to temporal and spatial variability in solute fluxes (Hayashi and Rosenberry 2002). It should be noted that groundwater exhibits a different geochemical composition than shallow subsurface flow that originates from precipitation events. Groundwater samples typically have a greater specific conductance (related to salinity), higher mineral content and elevated pH (Landmeyer 2011). Differences in chemical composition and biogeochemical activity are likely to manifest differences in pollutant removal from groundwater.

The hyporheic zone is the region beneath and alongside the streambed that represents the groundwater-surface water interface. While hydrologists have a solid understanding of most other hydrologic processes occurring in riparian areas, understanding of the exchanges between groundwater and in-channel surface waters is considerably less (UKEA 2009; Gu et al. 2012). Hyporheic flow patterns are multifaceted and influenced by the pressure gradient at the stream margin, (micro)morphology of the channel, the spatial distribution of hydraulic conductivity along the streambank, and the water exchange rate between the river and the riparian area (Kalbus et al. 2009; Lewandowski 2011). The complexity of the hydrologic regime is further compounded by seasonal variations, and at times intra-seasonal oscillations, in the direction of water flow (Wroblicky et al. 1998), as well as instances where ambient movement of deeper groundwater flows near the interface. Despite the confounding nature of the hyporheic zone, it is vital to consider hyporheic exchange when discussing subsurface fluxes of pollutants, as this zone is a direct link to streams through which contaminants can be transported to lotic systems (Gordon et al. 2004).

As groundwater approaches a streambank, biogeochemical activity increases. Groundwater flow rate and travel time through the riparian area influence natural chemical and biological processes that remove soluble pollutants.

Not only does the hyporheic zone play a critical role in regulating pollutant fluxes from riparian areas to the adjacent lotic environment, the heterogeneous structure at the interface, an oscillating water table, and shifting flow regimes exert a significant influence on biogeochemical cycling. The fluctuating water table alters oxygen concentrations and oxidation-reduction (redox) potential, and the flow regime provides a continuous source of reactants. This dynamic environment allows for anaerobic and aerobic processes to occur simultaneously and in close proximity (Wang et al. 2014). The strong hydraulic and biogeochemical gradients that result from mixing of the subsurface water coming from riparian areas and the surface water of the stream channel create biogeochemical hot spots that exhibit disproportionately high reaction rates compared to the surrounding matrix (McClain et al. 2003). For instance, the convergence of shallow subsurface flow and groundwater at the stream interface can generate hot spots of denitrification. Similarly, hot moments represent brief periods of elevated reaction rates relative to longer periods (e.g., during a rainfall event). Later sections of this chapter discuss this concept in more detail.

5.3.2. Soils

Soils play a critical role in influencing the transport and fate of pollutants in riparian areas. Soil texture and structure act as primary hydrologic regulators, significantly influencing the movement and distribution of contaminants. Comprised of a complex mixture of solid, liquid, and gaseous phases, riparian soils provide a heterogeneous matrix that supports plant and microbial communities. The solid phase encompasses mineral and organic particles of differing sizes and chemical composition that can sorb metals, organic chemicals, and other pollutants, thereby preventing them from migrating to the stream channel. The multiplicity and interaction of various biogeochemical processes in riparian soils contribute greatly to pollutant removal.

Unlike upland soils that are derived from terrestrial parent material, riparian soils primarily develop through the deposition of sediments that originate from various upstream sources. These alluvial deposits establish a heterogeneous patchwork of stratified sediments of different textures, which influences the movement of waterborne contaminants. Huggenberger et al. (1998) provide a good review of the geomorphology and characterization of alluvial soils, but a brief description is warranted here to provide a background related to the movement of contaminants through riparian soils. Deposits of clay, silt, sand, and gravel, and continual disturbance by seasonal high flows and occasional extreme flood events, create a complex mosaic of soil microenvironments that can vary considerably both vertically within the soil profile and along the length of the stream channel (Malanson 1993). A general trend can be observed where coarser sediments are deposited along steep-gradient streams while finer particles are washed downstream and deposited along low-gradient reaches as stream energy dissipates and wider floodplains develop (BLM 2003). Soil particle size also tends to decrease moving away from the stream edge (Huggenberger et al. 1998). This vertical, longitudinal, and lateral spatial structure results in disparate hydrologic conditions, and corresponding differences in pollutant fate, at locations in close proximity.

Rough-skinned Newt Taricha granulosa/*Renee Kinnick, WDFW*

The soil microenvironments described above influence hydrologic conditions, which consequently affect pollutant transport. Soil texture and structure dictate water storage capacity and retention, flow paths and hydraulic conductivity. Coarse textured soils enable a greater hydraulic conductivity that shortens hydraulic residence time in the riparian area, thereby reducing the period in which attenuation mechanisms can act on soluble contaminants before they reach the stream channel. Water may percolate below the root zone and limit potential interactions between vegetation and pollutants in deep, well-drained soils, such as those commonly found in forested riparian areas in humid climatic regions (NRC 2002). However, the root zone intercepts subsurface flow to a greater degree in forested riparian areas that exhibit shallow soils overlying a confining layer (e.g., shallow aquiclude). Overland flow that predominates in arid environments often undergoes limited infiltration due to a tendency for riparian soils to exhibit low permeability in these regions, which allows much of the runoff and associated waterborne contaminants to enter the stream with minimal attenuation (NRC 2002; Buffler et al. 2005).

Altered soil structure can influence pollutant attenuation (Vervoort et al. 1999). Compaction, for instance caused by over-grazing or vehicular traffic, and accompanying structural changes and surface soil instability can lead to diminished infiltration and amplified overland flow (Bohn and Buckhouse 1985; McInnis and McIver 2001). Soils with low infiltration rates typically exhibit a diminished capacity to retain pollutants in riparian areas (Johnson and Buffler 2008). Subsurface soil structure influences water movement and distribution in the unsaturated zone, affecting pollutant transport. Subsurface soil structure also impacts soil aeration and plant root growth, which have ramifications on pollutant fate. Preferential flow in soil macropores represents another facet of soil structure that affects pollutant removal (Allaire et al. 2015). Preferential flow of water through large connected void spaces in soils, which result from fractures, animal burrows and other processes, can enhance subsurface pollutant transport to river systems (Angier et al. 2002; Fuchs et al. 2009; Menichino et al. 2014). Alluvial deposits in riparian areas can further increase the connectivity of subsurface flows with stream channels, increasing lateral contaminant transport toward the stream channel. Preferential flow paths diminish the potential for contaminant sorption by both reducing interactions with soil particles and associated sorption sites and contributing to a higher flow velocity that reduces sorption kinetics. Although preferential flow paths have been a familiar concept in the field of soil science for some time, the recognized importance of these conduits has increased in recent years. This enhanced understanding is particularly important for controlling pollutants such as phosphorus that historically were not considered to undergo much, if any, subsurface transport (Angier et al. 2002; Fox et al. 2011a).

Artificial preferential flow also can be created by tile drains that are used by agricultural operations to remove excess water from the soil profile. Tile drains have been shown to enhance the movement of nutrients and herbicides from agricultural fields to streams, bypassing the natural attenuation processes that would have otherwise reduced the movement of these agrochemicals to aquatic ecosystems (Guan et al. 2011; Vidon and Cuadra 2011; Jaynes and Isenhart 2014).

In addition to playing an integral role in influencing the hydrologic linkage between upland areas and the stream channel, riparian soils provide a dynamic framework that regulates biogeochemical processes that act on contaminants. Soluble pollutants that have entered the soil profile, either from subsurface flow from upland areas or through infiltration in the riparian area, are subject to a variety of processes. Contaminants prone to sorption can be retained within the soil profile of the riparian area, reducing their potential of migrating to the stream channel. Chemicals typically have a greater affinity for clay minerals or organic matter, and thus the attenuation due to sorption will vary depending on

Riparian area along White River/Alan Bauer

the soil composition. Microbial assemblages in riparian soils can contribute to the assimilation, transformation, and degradation of a range of chemicals. Anaerobic conditions in saturated soils found in riparian areas differ considerably from the aerobic environments of adjacent uplands, altering the redox potential. The fate of numerous contaminants is influenced strongly by the redox potential of the soil and associated microbial communities. Fluctuating water tables can alter the redox state, resulting in seasonal variations in chemical species distributed within the riparian area. Although broad generalizations can be made, the literature highlights that temporal changes in soil complexities are not well understood (NRC 2002).

Vegetative communities are also impacted by soil conditions in riparian areas. Plant species exhibit habitat preferences that follow soil moisture gradients across riparian areas. The influence of vegetation on contaminant fate and transport will be discussed in the next section. However, it is salient to note here that soil compaction and other disturbances can lead to invasive plant species that can degrade riparian ecosystem function in relation to pollutant attenuation. Disturbed soils can lose their native species diversity and develop more homogenous stands of an exotic species (e.g., Himalayan Blackberry *Rubus armeniacus*) that often do not have the same vegetative density or root structure as the native population, thereby altering riparian area efficacy in pollutant removal.

5.3.3. Vegetation

Riparian vegetation also plays a critical role in attenuating pollutant transport to surface waters. Plants physically alter the movement of water and associated contaminants, provide habitat for microbial communities that assimilate and transform contaminants, and influence soil redox conditions that govern many biogeochemical cycles. Plant root systems provide bank stabilization, which reduces the direct introduction of sediments into aquatic ecosystems via streambank erosion. Soil stabilization also limits erosion and gully formation that can result from overland flow moving through a riparian area. In addition, forested riparian

Vegetated buffers reduce nonpoint source pollution to rivers and streams/U.S. Dept. of Agriculture, National Agriculture Imagery Program

areas have been shown to enhance instream processing of contaminants, including nutrients and pesticides (Sweeney et al. 2004). This section provides a brief overview of the influence vegetation has on pollutant attenuation in riparian areas and identifies critical points to consider when developing management plans. A number of focused review articles present further details on riparian vegetation-hydrology interactions and mechanisms by which plants can attenuate contaminants in riparian areas (e.g., Tabacchi et al. 2000; Williams and Scott 2009; Dosskey et al. 2010).

The high vegetation density inherent in streamside ecosystems serves a critical function in modulating overland flow and amplifying infiltration. The stems of grass and herbaceous groundcover, trunks of woody vegetation, and downed wood create resistance to overland flow, slowing it, and causing deposition onto and infiltration into the soil profile (Fisher and Binkley 2000; Schuster and Grismer 2004). Vegetation also helps maintain soil structure by increasing soil macropore formation through root growth and decay, which further enhances soil permeability. Soluble contaminants present in overland flow enter the soil column, augmenting attenuation through the various mechanisms discussed throughout this chapter. The reduced velocity of the overland flow caused by the riparian plant community causes suspended particles to settle out of the water column, reducing the transport of sediment and sediment-borne pollutants to the stream. Similarly, lower velocity flows will limit erosion within the riparian area, reducing potential contributions to stream sediment loads and avoiding gully formation, which concentrates overland flows and short circuits pollutant attenuation. A dense overstory canopy will further reduce erosion by intercepting precipitation

that would fall directly onto bare soil, and a dense plant community also reduces erosion of riparian soils by diminishing the velocity of runoff and floodwaters (Schultz et al. 2000).

Vegetation also increases soil surface roughness by producing herbaceous litter and woody debris, and by altering micro-topography, which slows surface flow and enhances infiltration of water. In contrast, sites with exposed mineral soil tend to exhibit low surface roughness that limits infiltration, and these sites are susceptible to increased transport to streams via overland flow.

Vegetation also influences the hydrologic behavior of subsurface and groundwater flow through streamside areas. Root uptake and transpiration by both herbaceous and woody plants can modify subsurface flow and soil moisture regimes in the unsaturated zone under unsaturated conditions. These alterations not only influence pollutant transport through riparian soils, but also affect redox conditions that govern biogeochemical transformations. Deeper-rooted plants can impact groundwater levels by reducing the pressure head at the water table through direct uptake or by extracting water along a water potential gradient. Hence, trees can cause daily or seasonal changes to water table levels, a phenomenon more prevalent in drier climates (Landmeyer 2011). Correspondingly, contaminant fluxes from riparian areas to surface waterbodies may exhibit similar diurnal and seasonal patterns. Water uptake by roots can similarly form a hydraulic barrier that creates an upward water flow through the soil profile that can reduce pollutant leaching and inhibit contaminant plumes from spreading horizontally (Pilon-Smits 2005).

Chinook salmon Oncorhynchus tshawytscha/*Morgan Bond*

While vegetation modulates water flow, it is vital to recognize the interconnected nature between riparian plant communities and hydrologic regimes. Riparian plant communities are determined largely by their adaptation to high moisture environments. Groundwater often represents a primary water source for riparian plants, principally trees and deeper-rooted flora. Consequently, alterations in groundwater regimes can have a detrimental impact on species diversity, density, and distribution, especially in drier regions where vegetation is typically concentrated along a narrow band of suitable soil moisture conditions (NRC 2002). Dramatic changes to plant communities can significantly limit the pollutant removal function of a riparian area. Thus, managers should consider the interrelationships between vegetation and subsurface hydrology. Kuglerová et al. (2014) expounded on this theme in a recent review examining the effects of forest management on groundwater processes and resultant water quality. The authors concluded that logging operations in riparian areas and hydrologically connected upland forests can alter groundwater regimes and affect water quality. They recommended site-specific riparian management where wider buffers would be implemented at critical groundwater discharge locations but narrower buffers would be allowed at sites lacking groundwater flow paths.

Riparian management strategies often emphasize the effectiveness of forested riparian areas for protecting water quality. However, research over the last two decades reveals that forests are not always the most effective vegetation for removing pollutants. Results can vary between sites, but research has shown that grass buffers (also called vegetative filter strips or grass filter strips) can yield similar reductions in sediments and sediment-borne pollutants (Correll 1997; Hawes and Smith 2004; Dosskey et al. 2010). Grasses and similar plants create a dense vegetative obstacle that slows overland flow, promotes deposition of sediments and attached pollutants, and provides an opportunity for the plants and associated microbes to capture soluble contaminants that enter the soil profile (Dillaha et al. 1989). Filter strips often serve as an effective alternative for protecting water quality from pollutants generated by upland activities where it is impractical to implement forested riparian areas, such as adjacent to agricultural fields, in urban settings, or along highways.

According to Lacas et al. (2005), much of the sediment removal effected by grass filter strips occurs mainly as a result of sedimentation immediately upslope of the strip in an area of still water that builds up against the strip's outer boundary. Furthermore, they note, the deposit formed upslope can expand into the filter strip, and that sediment deposition can increase to the point of forming channels that concentrate overland flow and induce the formation of water pathways with high velocities.

The primary drawback of a vegetative filter strip is that it addresses only pollutant removal and does not provide the other functions of forested riparian areas such as shading, large wood, and bank stability. Furthermore, grasses have a relatively shallow root structure and thus vegetative filter strips are not as effective in removing pollutants from groundwater. In contrast, trees inherently have a deeper root zone that can effectively intercept subsurface flow and often reach shallow groundwater. Therefore, forested riparian areas can be more effective at removing soluble pollutants in subsurface and groundwater flow through plant uptake and by associated microbial communities that act on the pollutants in the rhizosphere. Considering both situations, structurally diverse riparian areas that contain a combination of grasses, shrubs and trees provide a more robust system for mitigating a range of pollutants (Fischer and Fischenich 2000). A similar concept was proposed by Welsch (1991), who presented a three-zone model that considered an herbaceous filter strip between uplands and the start of the riparian area, followed by a managed forest, and culminating with an undisturbed forest immediately adjacent to the streambank.

Vegetation in riparian buffers also provides a range of phytoremediation processes, wherein plants and associated microbial populations remove, contain, degrade or eliminate pollutants. Investigations on the phytoremediation potential of riparian vegetation to attenuate nutrients have been conducted for some time and results are well documented. The management of vegetation specifically as a pollutant treatment technology for organic compounds and metals has gained acceptance in general over the last couple of decades (Pilons-Smits 2005), but studies on phytoremediation specific to riparian areas remains fairly limited. However, recent research suggests that the promotion of specific plant species to attenuate organic chemicals and metals in riparian areas can be used to treat localized areas of concern (e.g., chemical spills, industrial sites, mine tailings). Design of a phytoremediation treatment system depends largely on the three primary factors identified by Susarla et al. (2002) that govern contaminant uptake and distribution in plants and the surrounding soil system: physicochemical characteristics of the pollutant (e.g., polarity, lipophilicity), environmental conditions (e.g., soil moisture, pH), and plant characteristics (e.g., available enzymes, root structure). Specifics on the phytoremediation potential of different pollutant classes in riparian areas will be discussed later in the respective sections of this chapter, while more detailed descriptions of phytoremediation are readily found in a large number of scientific reviews (e.g., Salt et al. 1998; Pulford and Watson 2003; Pilon-Smits 2005; Sarma 2011).

5.4. Specific Pollutants

For the purposes of riparian area management, we are most interested in five types of water pollutants: sediments, excess nutrients (in particular, nitrogen and phosphorus containing compounds), metals (in particular copper), toxic organic compounds (especially pesticides), and pathogens (especially those related to agricultural animal waste). The physical, chemical, and biological processes through which riparian areas remove these pollutants from water vary tremendously. Consequently, the degree to which riparian areas can remove each of these pollutants, and thereby protect aquatic ecosystems, varies significantly. The following sections describe the complicated processes and mechanisms involved in determining removal efficacy of specific pollutants by riparian areas.

Vegetation in riparian buffers provides a range of phytoremediation processes, wherein plants and associated microbial populations remove, contain, degrade or eliminate pollutants.

5.4.1. Sediments

Sediment transport in streams is a natural process, but excessive sediment loads can significantly degrade water quality. The impact of sediments on aquatic organisms has been researched extensively for many decades, and can consist of lethal, sub-lethal and behavioral effects (Newcombe and MacDonald 1991). Sediments that settle out of the water column can smother gravel and cobble streambeds that are essential for salmonid spawning and habitat for benthic macroinvertebrates and other organisms that form critical links in stream food webs. For example, the filling of interstitial spaces in coarse sediment matrices with fine sediments are thought to reduce or eliminate refuge and foraging habitat for selected life stages of stream-dwelling amphibians (Welsh and Ollivier 1998). Increased turbidity levels, indicative of suspended particles, can elicit physiological stress in fish through gill abrasion, impaired immune function and lowered growth rates (Redding et al. 1987; Au et al. 2004). High suspended sediment loads reduce visibility, which limits fish foraging ability and influences predator avoidance reactions (Redding et al. 1987). Aquatic vegetation and periphyton can be reduced due to diminished light penetration. Sediments also impart indirect effects on aquatic organisms. Elevated turbidity decreases the water albedo (reflectiveness), which increases the fraction of radiant energy absorbed by the water and increases stream temperature, leading to corresponding reductions in dissolved oxygen concentrations. Furthermore, sediments serve as a transport mechanism for other pollutants, carrying attached contaminants from upland sources to the stream channel. Soil particles that harbor legacy contaminants perpetuate water quality problems, such as those caused by DDT in some Washington streams despite this pesticide being banned in 1972 (Ecology 2007).

Restored vegetation in riparian area to capture pollutants in runoff/U.S. Dept. of Agriculture, National Agriculture Imagery Program

While the extent of these detrimental effects is largely a function of concentration and duration, and there is little agreement on the degree of ecological risk in the literature, sediments are a ubiquitous pollutant that many scientists regard as the worst form of pollution in aquatic ecosystems (Newcombe and MacDonald 1991; Waters 1995).

Erosion rates in uplands can be significantly increased by agricultural activities, construction sites, deforestation, and other land uses. Riparian areas effectively remove soil particles carried from upland areas via overland flow by slowing overland flow velocities, promoting sediment settling, and acting as sediment sinks[5]. Mass wasting (also referred to as mass movement, mass failure, or slope movement), wherein large masses of soil and geologic material move downslope into a stream, acts as another potential sediment source in hilly or steep mountainous environments. Although limited in spatial distribution, this geomorphic process can overwhelm the sediment removal capacity of riparian areas.

Our understanding of the riparian area's sediment removal function will be enhanced with a basic understanding of sediment erosion and transport processes. Briefly, the impact of falling raindrops can loosen soil particles via splash erosion. Overland flow picks up soil particles and transports them downslope through sheet erosion. Generation of concentrated flow paths results in rill erosion, which enhances sediment delivery and increases erosion rates through flow that is on the order of a few centimeters deep. Gully erosion develops when deep, narrow channels form, leading to increased erosion and more rapid sediment delivery. It is important to prevent the formation of rill and gully erosion entering riparian areas because channel formation extending into riparian areas short-circuits sediment removal and represents a direct conduit from terrestrial to aquatic systems. Merritt et al. (2003) provide an extensive review of various erosion and sediment transport models that can be employed to determine the potential sediment loads that could enter a riparian area.

While the upland hydrologic regime acts as a principal driver governing the movement of eroded soil particles, the aforementioned erosion mechanisms can be significantly tempered by intact soil cover. This can be accomplished by vegetation overlying a dense cover of herbaceous litter and woody debris. Vegetation increases surface roughness which slows surface flow and enhances water infiltration, both of which promote the settling out of sediments (Figure 5.1). Muñoz-Carpena et al. (1999) showed that spacing between vegetation is the most important parameter in sediment attenuation in riparian buffers, although soil characteristics, sediment load, and slope also play prominent roles.

Agriculturally generated nitrogen continues to present an issue, and it is becoming increasingly apparent that smaller hobby farms and lawn fertilizers used in suburban and urban areas can lead to severely degraded water quality.

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[5] Streambank erosion, channel incision, and surface erosion in riparian areas present other potential sediment sources. As this chapter focuses on the removal of pollutants generated by upland activities, these geomorphological processes are outside the scope of this chapter. However, it should be noted that riparian vegetation can increase bank stability, thereby decreasing erosion along the stream channel (see Chapter 2).

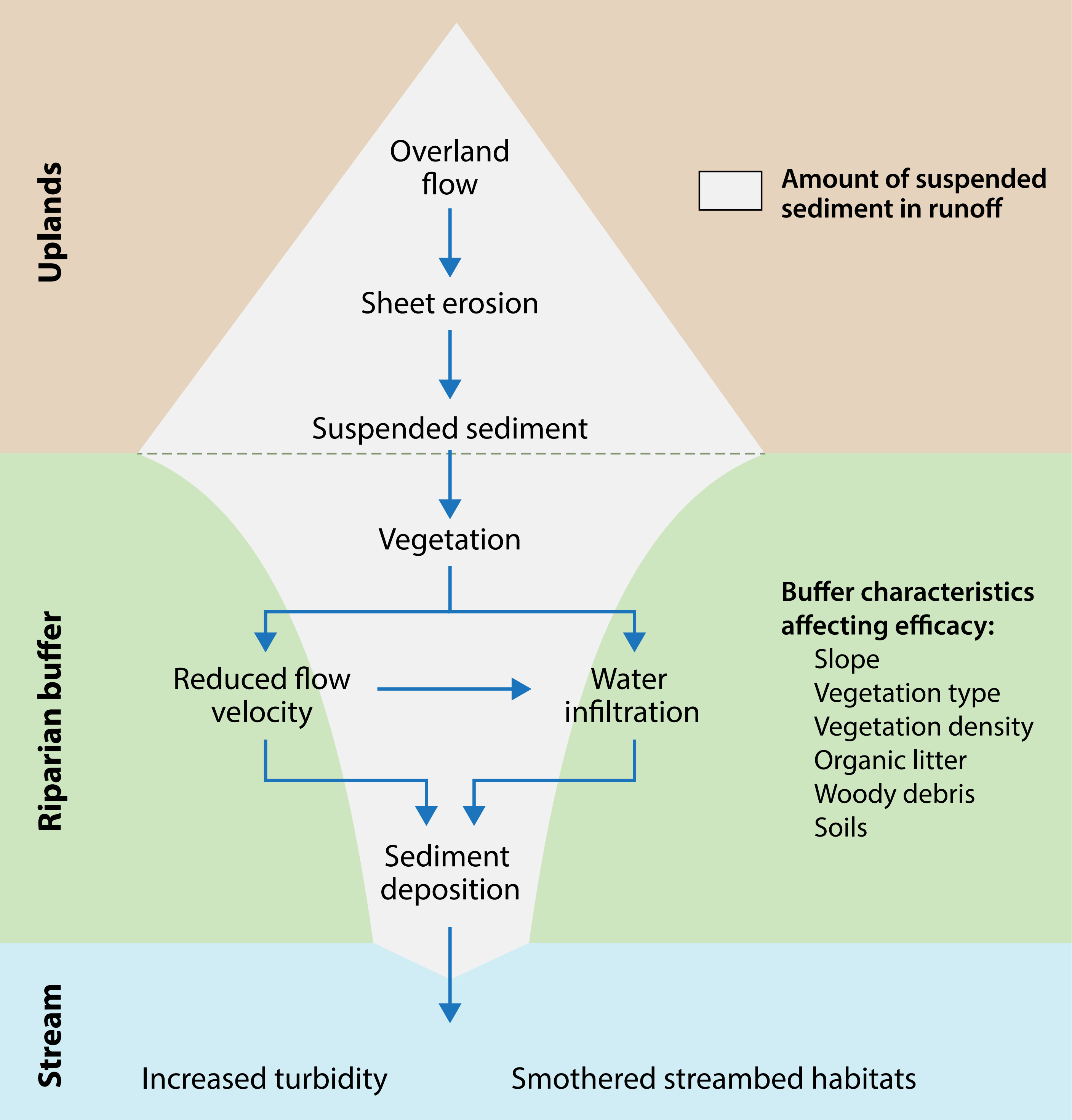

Figure 5.1 Efficacy of riparian buffer in removal of sediment from runoff. The amount of suspended sediment in runoff increases through sheet erosion of upland soils. Vegetation, woody debris, and organic litter in riparian buffer reduce flow velocity. Reduced flow velocity and increased water infiltration into soils causes deposition of suspended sediments. Sediments not removed from runoff increase turbidity and can smother gravel or cobble streambed habitats.

Riparian buffer widths required for sediment attenuation vary considerably depending on the site-specific conditions (e.g., slope, area of adjacent uplands, drainage paths, vegetation density). Sediment particle sizes are also a consideration in buffer width design – larger particles are deposited within the first 10 to 33 ft (3 to 10 m) of a buffer but smaller particles settle out at greater distances (Sheridan et al. 1999; Syversen et al. 2001; Dosskey et al. 2002; Yuan et al. 2009). The potential quantity of overland flow should also be a consideration when determining buffer widths, and hence, upland area should also factor into the determination of buffer width. Leeds et al. (1994) suggested that on agricultural lands the ratio of field drainage area to buffer area should be no greater than 50:1, and for consistently higher water quality the ratio should be on the order of 3:1 to 8:1. From research conducted on forested lands undergoing timber harvest, Clinton (2011) concluded that leaving a forested riparian buffer of 33 ft (10 m) would sufficiently reduce sediment transport to the adjacent stream, assuming responsible timber harvest practices, although greater widths have been recommended depending on the circumstance (Castelle et al. 1994). Furthermore, the relationship between buffer width and sediment attenuation is non-linear, necessitating a disproportionate increase in width to achieve an incremental increase in sediment removal. This relationship was described in a literature review of research on riparian buffers conducted by Sweeney and Newbold (2014). They concluded that a sediment removal efficiency of about 65% is typical for 30 ft (10 m) buffers but an 85% removal efficiency required more than twice that width, about 100 ft (30 m).

5.4.2. Excess Nutrients

Excess nutrients, in this context referring specifically to nitrogen and phosphorus, have long been recognized as significant water quality impairments. Eutrophication, the nutrient enrichment of aquatic systems, promotes algal blooms that shift community structure. Subsequent bloom die off leads to oxygen deficiency in the water column as oxygen is consumed by microorganisms during the decay of the excess organic matter. Hypoxia can have devastating consequences on aquatic animal communities, killing immobile organisms (e.g., clams) and mobile species (e.g., fish) that are not able to escape the deoxygenated zone. Excessive accumulation of cyanobacteria (formerly known as blue-green algae), resulting from shifts in nutrient ratios in the water, can release cyanotoxins which can cause dermal reactions, gastroenteritis, liver and kidney poisoning, and neurological impacts in humans or animals following water contact. Canine deaths due to cyanotoxin exposure have been confirmed throughout the country, with Washington reporting the highest number of cases among 27 states (Backer et al. 2013). In some cases, selected nutrients are toxic at concentrations within typical fertilizer application ranges. For example, embryos of the highly aquatic Oregon Spotted Frog *Rana pretiosa*, a federally threatened species in the Pacific Northwest, all died after 15-day exposure to nitrite (NO_2^-) concentrations lower than 2 mg/L (Marco et al. 1999).

Fertilizers and manures are major sources of excess nutrients entering riparian areas. Wastewater discharges and leaking septic tanks introduce nutrients to riverine systems, but those sources fall outside the scope of this review. Agricultural activities have long been blamed as the biggest nonpoint source for nutrients. The enormous contribution of phosphorus to the eutrophication of surface waters led to extensive promotion of various best management practices (BMPs)[6] for reducing off-field transport of phosphorus. Nevertheless, there has been a general lack of improvement in water quality of many receiving

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[6] In general usage "best management practice" (BMP) means one or more practices that are determined to be an effective and practicable means of preventing or reducing the amount of pollution generated by nonpoint sources to a level compatible with water quality goals. Under Washington State regulations, BMP means physical, structural, and/or managerial practices approved by the Washington Department of Ecology that, when used singularly or in combination, prevent or reduce pollutant discharges (WAC 173-201A-020).

streams. In addition to existing agricultural sources of phosphorus, it is increasingly evident that "legacy phosphorus" from past activities that accumulated in watersheds contributes to current water quality problems (Jarvie et al. 2013). Agriculturally generated nitrogen continues to present an issue, as it can leach through the soil and move with shallow groundwater flows through riparian soils and enter receiving waters. It is also becoming increasingly apparent that smaller hobby farms and lawn fertilizers used in suburban/urban areas can lead to severely degraded water quality (Cheng et al. 2014).

For a thorough discussion of nitrogen and phosphorus dynamics in natural, undisturbed riparian areas see Chapter 6. For this chapter it is salient to note the differing biogeochemical cycles exhibited by nitrogen and phosphorus: nitrogen displays complex transformations between various species including atmospheric forms whereas phosphorus undergoes relatively simple shifts between soluble, sorbed, precipitate and organic forms. Nitrogen and phosphorus are often casually grouped together as nutrients, but this discrepancy in chemical behavior leads to significantly different considerations when assessing their removal by riparian areas.

Nitrogen

Due to its soluble nature in its aqueous forms, a significant portion of the excess nitrogen entering streamside areas will enter riparian soils either through infiltration or via groundwater transport. Nitrogen then can undergo attenuation through a variety of mechanisms. It is widely accepted that denitrification acts as the primary pathway for removing nitrate (NO_3^-) from water in riparian areas (Fennessy and Cronk 1997; Cey et al. 1999; Wang et al. 2014). This process is governed by specific denitrifying bacteria that use nitrate during anaerobic respiration, reducing it to the gaseous forms of nitric oxide (NO) and nitrous oxide (N_2O) and ultimately molecular nitrogen (N_2). These gaseous species can then volatilize in harmless forms to the atmosphere and be completely eliminated as a potential pollutant. This process requires the presence of a shallow water table, anoxic conditions, denitrifying bacteria and an available carbon source—conditions commonly associated with riparian areas. Hydric soils, despite occupying a limited area within riparian areas, are important sites of anaerobic activity that promote denitrification. If groundwater bypasses riparian soils then nutrient pollution of surface waters may occur despite the presence of a riparian buffer (Burt et al. 1999). In a highly cited critical review of research on nitrate removal in riparian areas, Hill (1996) asserted that understanding denitrification requires a better understanding of groundwater interactions with riparian vegetation and soils.

Plant uptake of dissolved nitrogen in subsurface flows moving through riparian areas provides an additional mechanism that can remove considerable amounts of nitrate. A significant difference from denitrification though is that vegetative uptake is often viewed as transient storage for nitrogen because nitrogen assimilated by plants will be reintroduced to the ecosystem following leaf senescence or plant death. Both grasses and trees have been widely shown to remove nitrogen in riparian areas, but studies have yielded contradictory observations and conclusions regarding which vegetation type is more effective (Groffman et al. 1991; Osborne and Kovacic 1993; Schnaebel et al. 1997). A meta-analysis performed to assess the key riparian variables responsible for nitrogen removal concluded that forested buffers exhibited the greatest propensity for nitrogen uptake, and surpassed the efficacy of both grasses and communities composed of both trees and grass (Zhang et al. 2010). Although this study did not end the debate, many researchers believe that deeper and more expansive root structure allows trees to take up greater proportions of nitrogen. A riparian area's age may also be a variable affecting nitrogen removal. Cors and Tchon (2007) evaluated denitrification enzyme activity in two riparian areas of different ages: 6 and 20 years old. They found the older system was significantly more effective at denitrification, which was attributed to a greater accumulation of carbon.

Research focusing on hot spots and hot moments has provided a more nuanced understanding of nitrogen dynamics in riparian areas. Hot spots and hot moments that develop in the hyporheic zone are increasingly being recognized as a contributing mechanism to nitrogen attenuation (McClain et al. 2003; Vidon et al. 2010; Gu et al. 2012). A landmark paper by Hedin et al. (1998) exemplified the concept of enhanced nitrogen removal at hot spots along the soil-stream interface in a mixed forested-agricultural watershed. A first-order stream exhibited two converging primary flow paths, one consisting of shallow subsurface flow containing high concentrations of chemical reducers (i.e., dissolved organic carbon, CH_4, and NH_4^+) and the other a near-stream upwelling of deep groundwater containing high concentrations of oxidizers (i.e., NO_3^-, N_2O, and SO_4^{2-}). A zone of high denitrification resulted that was less than 3 ft (1 m) wide and remained active for the two-year study period. Hedin et al. (1998) suggested that nitrate removal via denitrification could be sustained depending on the spatial location of high flow pathways, the propensity of the chemical environment to support rapid denitrification, and the availability of oxidizable carbon. Furthermore, they recommended that management actions enhancing natural carbon inputs could be adopted to optimize nitrate removal by denitrification.

Phosphorus

Unlike nitrogen, phosphorus possesses a binding affinity for soil particles and typically enters riparian areas adsorbed to soil particles carried by overland flow. Hence, phosphorus removal is closely connected to sediment removal by riparian areas. A comprehensive review of phosphorus dynamics in grass buffer strips surmised that that optimum phosphorus retention occurs in filter strips that are at least 16 to 33 ft (5 to 10 m) wide (Dorioz et al. 2006), but this recommendation is debatable. It is generally recognized that a positive correlation exists between buffer width and phosphorus retention in riparian areas, and hence, buffers greater than 33 ft (10 m) offer greater removal efficacy. Independent studies have consistently found 60% to 80% phosphorus removal by buffers roughly 15 to 53 ft (4.6 to 16.3 m) wide (Dillaha et al. 1989; Vought et al. 1994; Lee et al. 2003). Forested riparian ecosystems assessments have yielded less consistent results, with studies showing phosphorus removal ranging from 30% to 80% in deciduous riparian areas (Lowrance et al. 1984; Cooper and Gilliam 1987). This discrepancy corresponds with the premise that a robust stand of grass provides better attenuation of sediments and associated pollutants than forested riparian areas.


Unknown species of crayfish in the Deschutes River/Steve Boessow, WDFW

Even though these results appear convincing at first glance, a significant proportion of the body of research consists of short-term studies that fail to account for changes in riparian function over time. While it can confidently be stated that riparian areas offer effective short-term control of sediment-bound phosphorus, uncertainty exists regarding their long-term efficacy. Filter strips receiving sediment loads are well known to become less effective over time due to internal erosion and sediment accumulation. The continual incorporation of phosphorus into surface soils presumably would lead to a saturation of the system that could not only decrease removal of new phosphorus entering the riparian area, but also lead to release and export of dissolved phosphorus to the stream that had previously been stored in riparian soils (Muscutt et al. 1993). Some studies have reported significant removal of soluble phosphorus in surface runoff (Peterjohn and Correll 1984). However, contradictory reports indicate little retention of soluble forms, with the majority moving through the area in surface runoff (Daniels and Gilliam 1996). Plants take up some of the phosphorus in the riparian soils, but this mechanism is not nearly as significant in removal as it is for nitrogen. Increased removal of soluble forms could potentially be achieved with various amendments, such as biochar (charcoal used as a soil amendment), that could enhance phosphorus sorption (Zhang et al. 2014).

A relatively new research topic related to phosphorus attenuation focuses on subsurface transport of soluble phosphorus forms. It was long accepted as dogma that phosphorus did not leach or move through the soil profile. Although scattered findings contradicted this view during the preceding decades (e.g., Peverill et al. 1977), this conviction remained recalcitrant and was still taught in university soil courses up until the last decade. This belief has now been dispelled by increasing evidence that phosphorus can move through the soil profile, particularly in coarse textured soils. A preponderance of the recent research on this topic has been conducted in upland soils, but the findings are relevant for riparian areas and are particularly true in coarse alluvial soils. This has been demonstrated by the rapid transport of soluble phosphorus forms through preferential flow paths (Fuchs et al. 2009). This movement can serve as an important transport mechanism that should be considered when determining potential phosphorus fluxes reaching the stream channel. Although extensive literature exists on phosphorus cycling in a variety of settings, relatively little information is available on phosphorus dynamics in the hyporheic zone and further research is needed to understand phosphorus flux through this interface between the stream and groundwater flowing through riparian areas (Zhang 2014).

While it can confidently be stated that riparian areas offer effective short-term control of sediment-bound phosphorus, uncertainty exists regarding their long-term efficacy. Riparian filter strips are well known to become less effective over time.

5.4.3. Metals

Metals (as well as metalloids that share similar properties, such as arsenic) are elements naturally found in geologic material. They enter the environment either through natural weathering processes or through anthropogenic activities. Localized metal deposits can result in elevated concentrations in streams that can be hazardous to biota. Metal pollution in aquatic ecosystems appeared as a serious environmental issue on a wide scale in the late 19th and early 20th century due to increasing industrial and mining activities. Contamination sources have grown over time to include a range of other activities, the most pertinent to this discussion on riparian areas being runoff from urban and agricultural lands. While mining activities present significant metal inputs into certain streams in Washington, protecting water quality from these sources requires the implementation of specific remediation techniques that are outside the scope of this chapter.

Metals present a significant environmental hazard that can lead to a variety of health issues in plants and animals. While some metals are essential elements required for normal growth and development of biota, these can also impart toxicological issues when concentrations exceed a given threshold. Among other health effects, metals can be carcinogenic, cause reproductive problems, and interfere with the normal function of the heart, bones, intestines, and kidneys. While debate continues over the specific mechanisms associated with metal toxicity in plants, it is generally recognized that oxidative stress represents a primary driver that results in cellular damage (Pinto et al. 2003). Some heavy metals bioaccumulate and biomagnify, which leads to elevated concentrations exhibited at higher trophic levels (Boening 2000). The propensity for metals to bioaccumulate can be magnified by biologically mediated transformation into organometallic complexes, such as methylmercury. Metal toxicity is often related to the chemical species formed, which often relates to their bioavailability. For instance, chromium(III) compounds are not considered a significant health risk while chromium(V) is a well-known carcinogen. Similarly, metal complexation with dissolved organic carbon often inhibits their bioavailability.

While copper's detrimental environmental impacts have been recognized for many decades (Sprague and Ramsay 1965), it has received increased attention in recent years as toxicity symptoms have become more pronounced. Copper is an essential mineral for organisms, contributing to enzymatic and metabolic function, but toxic effects have been observed at all levels of the aquatic food chain when it exceeds particular concentrations. Copper has been shown to have detrimental effects on plants, retarding growth at concentrations as low as 1.0 part per billion (ppb) and impeding photosynthesis at 5.0 ppb levels (USEPA 1980). While reductions in algal production can theoretically exert an indirect effect on zooplankton by limiting their food source, copper can impart a lethal effect on these organisms at low parts per billion levels (Ingersoll and Winner 1982). Similarly, significant mortality has been shown to occur in benthic macroinvertebrates at similar copper

Juvenile salmon need clean water/Ned Pittman, WDFW concentrations (Clements et al. 1992). Copper presents a litany of issues to fish. Lethal concentrations vary considerably between fish species and fluctuate depending on environmental conditions. Sub-lethal effects are conveyed at lower concentrations, which include hindering osmoregulation, weakening immune response, altering enzymatic and metabolic activity, impairing olfaction (sense of smell), altering migration patterns and predator avoidance behavior, and modifying hatch rates (Lortz and McPherson 1977; Sorensen 1991; Rougier et al. 1994; Baldwin et al. 2003; Monteiro et al 2005; Sandahl et al. 2007; McIntyre et al. 2012). Synergistic impacts that occur when copper is present in combination with other pollutants highlights the current concern over this element (Hecht 2007). Effects similar to those observed in fish have been recorded in amphibians following exposure to rather low concentrations of copper ions (Cu^{2+}). For example, the concentration of copper at which half of larval Northern Leopard Frogs *Rana pipiens* would die following a seven-day exposure was estimated at 0.067 mg/L (Redick and La Point 2004). The same study revealed lethargy, loss of equilibrium, and loss of appetite at copper concentrations of 36 ppb. A widespread pattern in copper exposure studies with amphibians is a general impairment of predator evasion behaviors (Garcia-Muñoz et al. 2011).

Anthropogenic copper inputs to the environment include those of other metals, but what sets copper apart are two significant sources. The first is the agricultural use of copper-based pesticides. Copper sulfate and other copper-based pesticides are frequently used as a fungicide to protect a variety of crops, including fruits and potatoes. The second is the release of copper from vehicle brake pads. This unique source has received considerable attention as an environmental hazard worldwide, and has led to the 2010 adoption of Washington State's Brake Material Friction Act (RCW 70.285) that is phasing out copper in automobile brakes (Straffelini et al. 2015). Copper can also enter the environment through atmospheric deposition, which is a potential pathway in industrial and mining regions (USEPA 1980). These sources introduce copper into the environment as a nonpoint source pollutant, which makes copper very different from other metals. While the following discussion will continue to refer to metals in general, the reader should recognize that the nonpoint source nature of copper makes it the primary metal of concern in Washington in regards to attenuation in riparian areas.

Metals form various inorganic and organic species, the nature of which directly relates to its bioavailability and mobility through a riparian area. Metal speciation is driven through complex biogeochemical processes. Environmental conditions and the chemical nature of the surrounding constituents play a significant role in dictating the form that a metal might take. This is particularly relevant to riparian areas where fluctuating water tables alter redox conditions, and correspondingly metal dynamics in riparian soils. Daily oscillations related to the photocycle, for instance, have been shown to occur in aquatic ecosystems as a result of biogeochemical interactions involving temperature, acidity, and biofilms (Nimick et al. 2011). Consequently, zinc, manganese, and cadmium can increase several-fold between late afternoon and early morning. In contrast, arsenic undergoes the opposite diel pattern; meanwhile copper and lead do not undergo regular diel cycling.

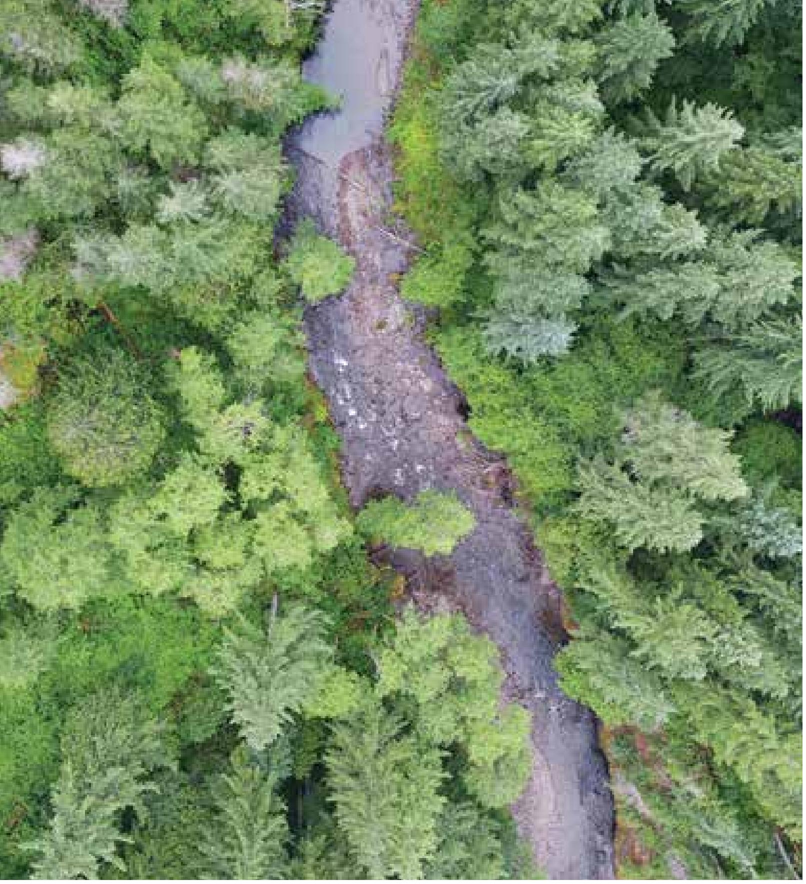
The metal species resulting from soil acidity and redox conditions can form precipitates, exhibit an affinity to clay minerals and oxide/hydroxide compounds, and thus be more readily retained in riparian areas. Conversely, metals stay in solution as more mobile forms that leach and move toward the stream. Typically, metals more readily enter solution under acidic conditions. Meanwhile the ionic strength of the water (a measure of the concentration of all the ions in solution) presents a confounding variable in these processes. The impact of other ions manifests through enhanced metal desorption from soil particles with increasing salinity. Carbonates in riparian soils promote metal retention by buffering acidity (i.e., preventing increased acidity), limiting metal solubility, and forming insoluble precipitates with metals.

Organic constituents in soil-water systems also play a major role in metal fate and transport processes. It has been clearly understood for some time that humic substances, a classification of complex and heterogeneous macromolecules that represent a substantial component of natural organic matter, have the capacity to bind significant amounts of metals (Reuter and Perdue 1977). While association of metal ions with soil organic matter can retain metals, humic substances can act as colloids and contribute to colloid-enhanced movement. The environmental factors and mechanisms involved in metal-humic interactions are multifaceted, but it is salient to recognize the impacts that organic matter in riparian soils can have on the movement of metals to the stream channel. A more detailed explanation of the inorganic and organic factors that influence metal behavior is presented in a review by Du Laing et al. (2009), which includes a specific focus on riparian areas.

Unlike organic contaminants that can degrade, metals are recalcitrant and removal in riparian areas depends largely on retention. Although the brief description of metal chemistry provided here elucidates the primary factors that affect their movement in the environment, management practices to enhance their attenuation are limited, and metal mobility will be driven overwhelmingly by environmental conditions. An exception to this statement is phytoremediation.

Phytoremediation of metals offers promise as a means to enhance metals removal in riparian areas. Although significant research on this process did not start in earnest until the late 1990s, since then hundreds of scientific papers have been published on the topic. Correspondingly, there is a plethora of review articles (e.g., Ghosh and Singh 2005; Sarma 2011; Koptsik 2014; Laghlimi et al. 2015). Four classes of phytoremediation can influence metal attenuation in soil, including phytoextraction (removal of metals from the soil-water system by plants), phytostabilzation (minimizing metal transport and sequestering them in the soil near the roots), phytovolatilization (uptake and transpiration by plants; this process is primarily limited to mercury and arsenic), and rhizofiltration (sorption, concentration and precipitation of metals in the root mass). Of these, the first two mechanisms are the most effective in attenuating metals (Laghlimi et al. 2015). In general, trees are a more effective form of vegetation due to their size, perennial nature, and the depth of root growth. Different metals will accumulate in different parts of a tree. For instance, copper tends to be immobilized and retained principally in the roots while zinc moves into the branches and leaf mass (Pulford and Watson 2003). Some plant and tree species act as hyper-accumulators that are tolerant of high metal concentrations in soils and can amass metals in their tissues through detoxification mechanisms (Sarma 2011). These species could be preferentially planted in areas where metals are a particular concern to help attenuate metal movement to the stream channel. As leaves will fall to the ground, riparian management could include the periodic removal of vegetative debris to off-site locations to avoid subsequent movement to the stream. Often there is no need to plant nonindigenous plants as a variety of common native riparian vegetative species can provide phytoremediation capabilities. For instance, willows (*Salix* spp.) have been shown to attenuate various metals in riparian areas to great effect (Bourret et al. 2009; Zhongmin et al. 2012).

Soil amendments offer another form of "green remediation" that has been increasingly used in soil systems since the turn of the century. A wide variety of amendments can be used either to mobilize metals by promoting uptake via phytoextraction or to immobilize (stabilize) metals in a solid phase. For instance, chelating agents have been applied at sites around the globe to enhance desorption, while phosphate


Tilton River/George Fornes, WDFW

fertilizers are common amendments intended to intensify metal sorption and form precipitates. Several topical reviews describing assorted soil additives have been published that provide more detail on this *in-situ* remediation technique (e.g., Bolan et al. 2014; Mahar et al. 2015). Similarly, over the last decade biochar has seen increased application in the context of attenuating metals (Ahmad et al. 2014). Although studies on the use of soil amendments to attenuate metals in riparian soils are limited, recent findings showed that additions of 11 different materials, such as biochar, bentonite, cement bypass kiln dust, and limestone to a floodplain soil could provide a potential remediation technique (Rinklebe and Shaheen 2015). An evaluation of the amendments in riparian soils heavily contaminated with copper (over 3,000 mg/kg) showed that most of the additives shifted the geochemical forms of copper through either increased sorption or by transformation to more plant available forms. Although the authors concluded that the low amendment rates that could practically be applied would inadequately immobilize copper, the increased copper concentration found in grass tissues at the same site indicates a means of enhanced phytoremediation. Clearly, the use of soil amendments requires further field verification in riparian areas, but this approach to enhancing natural attenuation mechanisms shows promise.

5.4.4. Pesticides and Other Organic Compounds

An extensive array of organic contaminants exists in the environment that can threaten water quality in aquatic ecosystems. This category of compounds describes molecules that contain carbon covalently bonded to other elements, and includes both naturally derived and synthetic chemicals. This chapter focuses on anthropogenic compounds that exert a toxic effect on aquatic organisms.

The multitude of organic chemicals makes it impractical to touch on each class here, but common examples that present environmental health risks in aquatic habitats include polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), and pesticides. While the detrimental effects of many of these pollutants have been recognized for some time, studies continually identify emerging contaminants of concern ("emerging" does not only connote new compounds, but also chemicals that have been in use without realizing the extent of their toxicity). For instance, hazards associated with flame retardants, such as polybrominated diphenyl ethers (PBDEs), have been highlighted over the last decade (Covaci et al. 2011; Ye et al. 2015). Recent research emphasis also has been placed on a variety of trace organic compounds, including pharmaceuticals and personal-care products (PPCPs), as it is now recognized that these chemicals can exert negative impacts on fish and other aquatic organisms at both the individual and population levels even at extremely low (e.g. parts per trillions) concentrations (Mueller 2004; Kümmerer 2009a; Kümmerer 2009b; Luo et al. 2014). In addition, there is growing awareness that while individual pesticides may have little or no effect when examined alone, that same pesticide as part of a pesticide mixture may be lethal to amphibians (Hua and Relyea 2014).

While only a brief discussion was provided above on the various organic pollutants that act as environmental contaminants, a little more information on pesticides is warranted, as they are more likely to be used in uplands and transported into riparian areas. Pesticides consist of any substance that is used to control fungal, plant, or animal pests. These chemicals represent a broad range of different chemicals that are derived from various sources such as plant extracts (nicotine, pyrethrum, neem oil), inorganics (e.g., copper sulphate; inorganic pesticides will not be discussed here), and synthetics. Synthetic pesticides are themselves a broad class that includes legacy organochlorines (chlorinated hydrocarbons such as DDT), organophosphates (e.g., Malathion), phenoxyacetic acids (e.g., MCP), carbamates (e.g., carbaryl), and neonicotinoids (e.g., imidacloprid). Pesticide pollution of streams can impact non-target organisms resulting in a wide variety of physiological and ecological problems (Mann et al. 2009), including altered growth (Alvarez and Fuiman 2005) and reproduction (Moore et al. 2007), altered community dynamics (Fleeger et al. 2003), and the loss of species (Relyea and Diecks 2008).

The ecological impacts of organic pollutants on aquatic organisms vary considerably, depending on not only the contaminant type but also the environmental conditions. Various compounds can be carcinogenic (causing cancer), mutagenic (altering genetic material and fomenting mutations), and teratogenic (stimulating abnormalities). Organic pollutants can also act as endocrine disrupting chemicals (EDCs), adversely impacting development, reproduction, neurologic, and immune system function in humans and wildlife (Colborn et al. 1993). Endocrine disruption has emerged as a key issue as studies have identified more cases of detrimental impacts, such as the discovery of atrazine-induced feminization of male Northern Leopard Frogs at very low concentrations (Hayes et al. 2003). This finding is controversial, but it has increased awareness that many organic compounds have potentially unrecognized endocrine-disrupting effects. Some organic contaminants, such as halogenated hydrocarbons, are capable of bioaccumulating in individuals and biomagnify up the food chain (Rahman et al. 2001).

To further complicate potential risk, pesticides rarely occur singly in aquatic environments. For example, a comprehensive decadal assessment of 178 streams in agricultural, urban, and mixed-land use watersheds found two or more pesticides more than 90% of the time and ten or more pesticides 20% of the time (USGS 2006). Ecological and toxicological research is increasingly recognizing the complexity of contaminant mixtures, which may have antagonistic, additive, or synergistic interactions in biota (Cedergreen 2014). For example, combining the pesticides diazinon and Malathion inhibits brain activity, and therefore swimming performance in fish at concentrations more than 50 times lower than the concentrations that cause an impact when these pesticides are used singly (Laetz et al. 2013). Although the examples given focused on pesticides, mixtures of hydrocarbons originating from roadways, parking lots and other locations can also be found in urban watersheds, including pollutants such as PAHs that originate from coal-tar-sealed parking lots (Scoggins et al. 2007). Amalgamations of these urban generated organic contaminants have been shown to impart both sub-lethal and lethal effects on aquatic organisms and disrupt stream communities (PSAT 2007; Scoggins et al. 2007; Spromberg et al. 2016).

The sheer number of different sub-classes of pesticides makes it impractical to provide information on the chemical behavior of all the potential contaminants that might be encountered in riparian areas. Pesticide pollution is extremely complex due to the multitude of types, the wide range of molecular sizes, the diversity of chemical structures, and the ability of these compounds to react and transform into new products. However, some basic information on organic compounds will enable a better understanding of the effects riparian buffer size and structure have on pesticide pollutants. It is essential to identify the physicochemical properties of organic contaminants to adequately assess fate and transport pathways through riparian areas. Specific gravity, water solubility, vapor pressure, and partitioning coefficients such as the octanol/water partition (log K_{OW}) and organic carbon partition coefficients (K_{OC}) all serve as predictors of a substance's potential mobility and attenuation in riparian areas.

Natural organic matter has a significant impact on the degree of attenuation riparian soils impart on pesticides. The aforementioned K_{OC} represents the ratio of the amount of chemical adsorbed to the soil to the amount of chemical that remains in solution, related to the mass fraction of organic matter in the soil (Delle Site 2001). This value essentially indicates the affinity of a pollutant to soil organic matter and is a predictor of its potential mobility through the soil, where higher K_{OC} values correlate to less mobile organic contaminants and low K_{OC} values correlate to more mobile organic contaminants. Hydrophobic, nonpolar compounds, such as PAHs, are typically associated with particles in soil-water systems, whereas polar contaminants often exist in dissolved forms that exhibit a greater propensity for movement through the ecosystem.

The relationship between pesticides and their respective K_{OC} values can be confounded by humic substances, which are a heterogeneous mixture of chemically complex macromolecules that comprise a substantial component of natural organic matter. Humic substances have long been known to exert a significant impact on the movement of organic contaminants (Hayes 1970; Khan 1972). These amorphous compounds, which are classified into humic acid, fulvic acid, and humin fractions based on their molecular size and chemical behavior, can effectively bind organic pollutants and present either water-insoluble entities that are retained in the soil column or water-soluble colloids that can facilitate transport through riparian soils to the stream channel. Even when present in small quantities, these chemically reactive compounds can dramatically influence the binding, persistence, and translocation of organic pollutants. The interactions between humic substances and organic contaminants are beyond the scope of this chapter, but their potential to affect K_{OC} values by up to an order of magnitude is salient to predicting their attenuation by riparian buffers (Grathwohl 1990; Kopinke et al. 2001). Nevertheless, in general the attenuation of organic pollutants in riparian areas is positively correlated with soil organic matter.

Pesticides and other organic pollutants can also be retained in riparian areas through sorption to mineral components in the soil, although often to a lesser extent than to organic matter (Delle Site 2001). The surface area of soil particles enhances the binding of organic pollutants, which increases with finer soil textures such as those found in clays. The mineral composition of the soil particles also influences binding affinity by offering different forms of attachment sites. Those compounds that do not attach to soil material are susceptible to leaching, which can result in them entering the groundwater and being transported to the stream. Contaminants associated with inorganic particulates are usually much less bioavailable, although they can become bioavailable if particles are ingested by biota (Moermond et al. 2004).

Pesticides and other organic pollutants can undergo abiotic and biotic transformations that further compound the complexity of their chemistry. Abiotic processes include degradation due to chemical reactions such as hydrolysis (Mitchell et al. 2014; Mitchell et al. 2015). Biotic transformations include biologically mediated reactions, such as bacterial metabolism. Although the potential for biodegradation is dependent on the chemical structure of the contaminant, indigenous soil bacteria can use many organic compounds as a metabolic substrate. Repeated exposure to the same type of pesticides can foster microbial communities that become contaminant specialists (Vidon et al. 2010). For example, soils receiving repeated applications of the pesticide 2,4-D harbor more 2,4-D-degrading microbes than similar plots receiving fewer applications (Gonod et al. 2006). Similarly, PAH-degrading bacteria were more abundant in rain gardens receiving stormwater runoff than upland control plots (LeFevre et al. 2012). The robust microbial communities found in riparian areas, resulting from conditions discussed previously, helps attenuate many organic compounds in riparian soils. Being biologically mediated, however, the degradation rates will fluctuate seasonally depending on temperature and soil moisture.

Phytoremediation by riparian vegetation provides an additional attenuation mechanism for organic contaminants. Although phytoremediation has received significant interest over the last two decades, the literature contains few papers on this method specific to pesticides in riparian areas. However, the principles are the same as those described in the vegetation section found earlier in this chapter, and presumably the results would be similarly positive. This premise was highlighted in a review article on the potential effectiveness of phytoremediation in riparian areas, where Karthikeyan et al. (2004) drew on similarities between streamside areas and other locations where phytoremediation has been effectively employed to treat pesticide-contaminated soil and water. Similarly, another review paper summarized a range of studies that investigated phytoremediation in aquatic settings, addressing plant species such as cattail (*Typha* spp.) and Cutgrass *Leersia oryzoides* which can be found along the riparian-aquatic ecosystem transition (Moore et al. 2011). A study directly examining the use of phytoremediation in riparian areas found that Tussock Sedge *Carex stricta*, Switchgrass *Panicum virgatum* and Gamagrass *Tripsacum dactyloides* led to approximately 70% reduction in total petroleum hydrocarbons under controlled riparian conditions, while there was only about 20% reduction associated with a Narrowleaf Willow *Salix exigua* and poplar (*Populus* spp.) (Euliss 2008). Following the same procedure in an uncontrolled riparian field site did not reproduce these promising results, but the authors presumed there was a continuous pollutant source re-contaminating the site during the study. A field study examining the phytoremediation potential of willows (*Salix* spp.) in riparian areas yielded a 49% decrease in PCB concentrations before contaminated groundwater reached the stream margin (Skłodowski et al. 2014). Although there is not enough information available in the literature to provide design guidelines for implementing a phytoremediation-based system, there is promise in this approach to attenuating organic contaminants in riparian areas and further research should be encouraged.

Mycoremediation, the degradation of contaminants by fungi, is a type of bioremediation that occurs in soils. This process likely takes place in riparian soils with high organic matter content where lignin-degrading fungi (more commonly known as wood-decay fungi, a class that includes brown rot, soft rot, and white rot) exist. These microorganisms possess the ability to breakdown organic contaminants, particularly PAHs, through the release of a variety of oxidizing enzymes (Haritash and Kaushik 2001). Research on promoting lignin-degrading fungi as a remediation agent started to appear in

The central problem faced by resource managers is determining the adequate riparian buffer width, composition, and structure to protect water quality with high degrees of efficacy, efficiency, and certainty.

Spawning Chum Salmon Oncorhynchus keta/*Ned Pitman, WDFW*

the literature in the 1990s, but did not receive much attention until after the turn of the century (Lamar et al. 1993; Boonchan et al. 2000; Canet et al. 2001). Results vary considerably, but it is not uncommon to observe 50% to 90% reductions in organic pollutants resulting from mycoremediation-mediated degradation (Lamar et al. 1993; Boonchan et al. 2000; Haritash and Kaushik 2001). Although this degradation process is mentioned here to highlight the mechanisms affecting organic pollutants in riparian soils, it has been suggested that mycoremediation could potentially be capitalized on as an engineered mitigation technique in riparian areas, but more research is needed (Jones 2009).

Despite the large body of knowledge that has been developed, the mechanistic understanding of pesticide transport through riparian areas is incomplete. The following summarizes our understanding of the typical transport pathways of a pesticide; this is essential information to consider when determining appropriate riparian widths for attenuation. Pesticide migration from terrestrial sources to stream channels can follow a number of different pathways. The specific transport mechanism governing pesticide movement from uplands through riparian areas and into surface waters is highly correlated to the type of pesticide in question. Surface runoff can carry pesticides off fields as either dissolved forms or attached to eroded soil particles. Leaching of compounds entails their vertical movement through the soil, after which they can either move toward the stream through the unsaturated zone or via groundwater flow. The chemicals that move in the dissolved form tend to be those that exhibit low to moderate sorption properties, as compounds that possess lower partition coefficients are prone to leach into the soil profile. Leaching is further promoted in regions with high precipitation and in soils that are coarse-textured, contain a low percentage of organic matter, or demonstrate a high degree of macropore flow. Pesticides that predominantly follow sediment-facilitated transport are generally those that display a strong affinity to soil particles (i.e., $K_{OC} > 1,000$ L/kg).

While the literature lacks information on the potential attenuation in riparian areas for most organic pollutants, there are a substantial number of papers on the movement of pesticides through buffer strips. A number of review papers evaluate pesticide attenuation in riparian areas, with a particular emphasis on managed vegetative filter strips (Krutz et al. 2005; Lacas et al.

2005; Reichenberger et al. 2007). Due to the complexity of the attenuation mechanisms involved, the removal rates within buffers vary considerably between disparate compounds, as well as when comparing the removal rates of the same pesticide at different locations. Depending on the conditions, mass removal rates for some pesticides can be negligible while others can reach 100%. Research has shown that herbicide type and filter strip width are primary factors affecting herbicide retention in riparian areas, while antecedent moisture conditions exert an additional effect wherein pesticide attenuation is generally negatively correlated with soil moisture. The type of vegetation appears to have negligible impact on pesticide retention, with studies showing that a grass-shrub-tree mixture exhibited comparable mitigation as grass only. Interestingly, one study found that pesticide mitigation was greater at higher incoming concentrations, which was attributed to better sorption to soil and plant material (Misra et al. 1996).

Due to confounding and often conflicting results, after 35 years of research, there remains a lack of consensus on the minimum filter strip widths needed for effective pesticide removal from surface and surface overland flow. The NRCS recommends a standard width of 49 ft (15 m) with the aim of achieving 50% effectiveness, but there is not much evidence to support this guidance (NRCS 2000). Another general recommendation for diffuse runoff suggests riparian widths of 33 and 66 ft (10 and 20 m) for hillslopes of under and over 328 ft (100 m) in length, respectively (Lacas et al. 2005). However, because this recommendation is based on only one experimental study, it may not be applicable to other locations with different hydrological, soil, and vegetative conditions.

Pesticide Spray Drift

Pesticide spray drift, the aerial transport of pesticide droplets off target areas during spray application, presents another pathway by which these chemicals can enter aquatic ecosystems. Unlike hydrological transport, pesticide physicochemical properties play a negligible role in aerial transport (although formulation does act as an influencing factor). Weather conditions (e.g., wind speed, temperature, and humidity), equipment type, application practices, and target crop all affect aerial transport. There is a better understanding of pesticide aerial drift to streams than that of hydrological pathways, either as a result of occurring in a less complex matrix, or as Reichenberger et al. (2007) asserts, due to aerial drift receiving more regulatory scrutiny.

Management of riparian areas can significantly reduce pesticide spray drift in many cases, particularly when conducted in conjunction with proper application practices. However, in their review of mitigation via windbreaks, Ucar and Hall (2001) found no general consensus on optimum riparian buffer widths to protect water quality. Their review reports that research has yielded buffer width recommendations that range from 20 to 150 ft (6 to 45 m) for ground application and 80 to over 4,000 ft (25 to over 1,200 m) for aerial application. Despite this lack of harmony in recommendations, individual cases have yielded positive results. Analysis of a forested riparian area in British Columbia found that a 25 ft (7.6 m) buffer limited the amount of herbicide reaching the stream to less than 0.1% of the aerial application (Feng et al. 1990). In an agricultural setting, a simple 10 ft (3 m) no-spray buffer reduced herbicide drift deposition to adjacent surface waters by 95%, and no measurable drift reached the stream at a width of 20 ft (6 m) as long as the wind-speed was below 15 mph (25 kph) in the direction towards the buffer (De Snoo and De Wit 1998). Many factors contribute to the effectiveness in reducing pesticide drift to acceptable levels. While wind-speed and application method exert significant impacts, vegetation height, density, orientation, and species composition all play pivotal roles. For instance, Ucar and Hall (2001) suggest that, in general, evergreen species display capture efficiencies 2 to 4 times greater than deciduous trees.

5.4.5. Pathogens

Pathogens impair more waterbodies in Washington than any other contaminant (USEPA 2015). Fecal contamination of surface waters can lead to the introduction of various pathogenic bacteria, fungi, viruses, protozoans, and worms that infect the gastrointestinal tract and present a health risk to both humans and wildlife. Common infections related to fecal pollution result from the presence of pathogenic bacteria such as *Escherichia coli* (*E. coli*), *Salmonella* spp. and *Vibrio* spp. or viruses such as Rotavirus. Primary pathogenic protozoans of concern include *Cryptosporidium parvum* and *Giardia lamblia,* which are able to form oocysts (protective spores) that allow them to survive outside of a host for long periods. Helminths represent a general class of parasitic worms, including flatworms and roundworms, which can act as intestinal parasites. The myriad pathogens that can be found in riverine systems following fecal contamination makes it impractical to list them here, but it is important to understand the four different pathogenic categories that can be conveyed to people through surface water contact or ingestion.

Pathogens comprise a diverse range of microorganisms that are typically found in low concentrations, making it impractical to test for specific pathogens when conducting water quality assessments. As an alternative, "pathogen indicator organisms" are used as a proxy to assess the potential presence of pathogens in a water sample. Commonly employed pathogen indicators include total coliforms, fecal coliforms, and *E. coli.* Total coliforms comprise a group of five genera of bacteria that easily can be analytically differentiated from other microorganisms, and while some occur naturally in soils, many are found in the gastrointestinal tract of warm-blooded animals and can serve as a general marker for fecal contamination. Fecal coliforms represent a sub-group of total coliforms that more specifically reside in the intestines of warm-blooded animals, making them a more accurate indication of the presence of fecal material in water. Washington water quality standards use fecal coliforms as indicator organisms to determine water quality impairment. *E. coli,* a species in the fecal coliform group, is a dominant inhabitant in the digestive tract and is generally considered the best indicator of fecal pollution of the three. As *E. coli* are indigenous enteric bacteria, they are not in themselves pathogenic although some strains are (e.g., *E. coli* O157:57). Although most of these indicator organisms are not pathogenic, it is common to refer to them as pathogens when discussing water quality (shortening the phrase "pathogens and pathogenic indicator organisms"), and while technically inaccurate, we will use that term in this section. We use the term in a way that should allow the reader to differentiate between whether we are referencing pathogens specifically or the broader group that includes pathogen indicator organisms.

As the pathogens in question are enteric, they are introduced into the environment through the feces of warm-blooded animals. Principal pathways for pathogens entering aquatic ecosystems are through direct defecation by livestock, wildlife, and pets in the riparian area and via overland flow carrying fecal material from upland pastures, animal feeding operations, fields that have received land-applied manures, or seepage from septic or sewage systems. Feces from domesticated dogs can be a predominant pathogen source in urban areas, as a study in Seattle attributed 30% of the fecal contamination to canines (Tobiason et al. 2002). Hobby farms also present an often-overlooked source adversely affecting water quality, and this has been shown to be an issue in Washington (Morace and McKenzie 2002).

The fate of pathogenic microorganisms in the environment is still not well understood in general and much uncertainty remains in the context of riparian areas. While many of the pathogens that can contaminate surface waters are obligate parasites, some can persist outside of an animal's gastrointestinal tract in the environment. The length of time these microorganisms can survive under different conditions is a current topic of debate in the research community. The general assumption in the past was that these enteric organisms were not able to persist in the environment for more than a few days following excretion in feces. However, more recent research indicates that bacterial pathogens can survive for much longer periods, and it is generally recognized that persistence in the environment for two to three months is not uncommon. Longer survival rates are possible given the correct conditions, as demonstrated in a laboratory study where *E. coli* O157:H7 persisted for over 190 days in manure-amended soils (Jiang et al. 2002). General factors that influence the survival of pathogenic bacteria that enter riparian areas include soil moisture, temperature, pH, sunlight, nutrient availability, soil texture, and biological interactions. Moist, nutrient-laden, shady conditions such as those often found in riparian areas theoretically would provide an environment in which pathogens could survive longer, potentially being picked up by overland flow and introduced to adjacent stream channels.

Although considerable research has been conducted through the years on bacterial movement through environmental systems, research is in the nascent stages of examining the transport and fate of pathogens across the landscape and in riparian areas. Unc and Goss (2004) offer a concise review on the transport of bacteria from manure in relation to protecting water resources, and although the material presented does not focus on streamside areas it does provide fundamental information on the mechanisms involved in pathogen transport. The body of research that has examined pathogen attenuation in riparian areas is small although expanding, and almost all findings are related to vegetative filter strips and do not consider natural riparian ecosystems.

The capacity of vegetative filter strips to remove pathogens is not well established and research results have demonstrated that these systems exhibit great variability in their effectiveness in removing pathogens. A number of studies have shown that filter strips impart no significant influence on pathogen and pathogenic indicator organism concentrations moving toward a stream channel (Walker et al. 1990; Schellinger and Clausen 1992; Chaubey et al. 1995; Coyne at al. 1995; Entry et al. 2000). Contradictory findings have been generated by other studies that promote the idea that well maintained filter strips can serve as an effective BMP to prevent pathogens from reaching surface waterbodies (Coyne et al. 1998; Fajardo et al. 2001; Tate et al. 2006).

Inconsistent findings result from a number of factors. Soil infiltration, antecedent soil moisture conditions prior to a rain event, vegetation status, topography, and rainfall intensity and duration all have been identified as key controlling variables (Muñoz-Carpena et al. 1999; Guber et al. 2009). Each of these parameters relates to the concept that pathogens being carried in overland flow are removed by their infiltration into the soil where they typically are filtered or adsorbed to soil particles. High intensity rain events and concentrated flow can significantly reduce pathogen removal efficacy below that which would occur under low-flow conditions (Fox et al. 2011b). Vegetative filter strip width plays a significant role in pollutant removal, where a linear

Although considerable research has been conducted on bacterial movement through environmental systems, research is in the nascent stages of examining the transport and fate of pathogens across the landscape and in riparian areas.

decrease in pathogenic bacteria has been consistently observed with increasing buffer width (Young et al. 1980). However, a potential exists for trapped pathogens to later be remobilized in subsequent runoff events. Higher concentrations of pathogens have been recorded in runoff during a second runoff event, which suggests that filter strips could become indirect sources of pathogens under some conditions (Collins et al. 2004).

5.5. Effect of Riparian Buffer Width on Pollutant Removal

The pollutant removal function of riparian areas has been studied for at least 40 years (e.g., Doyle et al. 1975), and the enormous quantity of scientific research over the past four decades has motivated numerous reviews of the scientific literature (Norris 1993; Osborne and Kovacic 1993; Barling and Moore 1994; Vought et al. 1995; Fennesy and Cronk 1997; Lyons et al. 2000; Dosskey 2001; Hickey and Doran 2004; Lacas et al. 2005; Krutz et al. 2005; Polyakov et al. 2005; Hoffman et al. 2009; Dosskey et al. 2010). Despite the large quantity of research and number of literature reviews, no widely accepted recommendations have emerged on minimum buffer widths needed to protect water quality. The lack of agreement amongst scientists is due, in part, to the surprising complexity of the mechanisms that remove pollutants from surface and subsurface flows in riparian areas, to the variety of research methods used to study pollutant removal by riparian buffers, and to the many different environmental conditions at research sites.

The quantitative metric of pollutant removal by riparian areas is referred to as removal efficacy, removal efficiency, or percent removal. These measures are usually defined as:

$$E_R = \frac{m_{in} - m_{out}}{m_{in}} \times 100 \qquad (1)$$

where m_{in} and m_{out} are the mass or concentration of the pollutant entering and exiting a riparian buffer, respectively. The primary question addressed by nearly all research on the pollutant removal function of riparian areas, and by nearly all literature reviews, is how does riparian buffer width affect removal efficacy? The main answers are: 1) removal efficacy increases as buffer width increases, 2) topographic slope and vegetation type strongly affect removal efficacy, and 3) the relationship between removal efficacy and buffer width is highly variable.

In addition to the many literature reviews cited above, several statistical meta-analyses of research results (Mayer et al. 2007; Liu et al. 2008; Yuan et al. 2009; Zhang et al. 2010; Sweeney and Newbold 2014) provide synoptic perspectives on what is currently known about the relationship between buffer width and removal efficacy (Table 5.1). Research results from throughout the world were collected by all five meta-analyses, but the majority were from North America. The remainder of this section discusses the findings of those meta-analyses.

Four meta-analyses looked at the results of research on sediment removal. The meta-analyses found that buffer width alone explained 28% to 37% of the variance in sediment removal results. Models for removal efficacy versus buffer width predicted a wide range of buffer widths necessary for 90% efficacy: 33, 39, 75, and 170 ft (10, 12, 23, and 52 m) (Figure 5.2). Three of the meta-analyses found that adding slope as an independent variable significantly improved the fit of their models to the data. Two meta-analyses (Liu et al. 2008; Zhang et al. 2010) using the same studies but different modeling assumptions, found that optimum slope for sediment removal was about 10%. In other words, removal efficacy increased as slope increased from 1 to 10%, but then removal efficacy decreased as slope increased above 10%. However, extrapolating these findings to other locations should be done cautiously as this relationship may not hold under many conditions. Zhang et al. (2010) found that buffer width, slope, and vegetation type together could explain 65% of the variance in sediment removal results. Curiously, they also found that buffers composed of grasses only or trees only were more efficacious at removing sediment than buffers composed of both grasses and trees (Figure 5.4).

Three separate meta-analyses looked at the results of research on nitrogen removal. The meta-analyses of Zhang et al. (2010) and Mayer at al. (2007) produced very different relationships for removal efficacy versus buffer width. Zhang et al. (2010) found that buffer width alone explained 44% of the variance in nitrogen removal results, but Mayer et al. (2007) found buffer width alone explained only 9%. Furthermore, Zhang et al. (2010) predicted that buffer widths of 80 ft (24 m) are necessary for 90% removal efficacy, while Mayer et al. (2007) predicted buffer widths of 436 ft (133 m) for the same efficacy (Figure 5.3). Separate models by Mayer et al. (2007) for surface and subsurface flows found a significant relationship (assuming $\alpha = 0.05$) between buffer width and removal efficacy for surface flows ($R^2 = 0.21$, $P = 0.03$) but not for subsurface flows ($R^2 = 0.02$, $P = 0.3$). Despite this result, Mayer et al. (2007) found that riparian buffers were more efficacious in removing nitrogen from subsurface waters than from surface waters. Mayer et al. (2007) found no significant relationship between nitrogen removal and buffer vegetation type but their collection of studies did show that buffers lacking trees (herbaceous cover only) were the least efficacious at removing nitrogen.

Zhang et al. (2010) found a relationship between vegetation type and nitrogen removal and that buffers composed of trees remove more nitrogen than buffers completely or partially composed of grasses. Zhang et al.'s model that included buffer width and vegetation type explained 49% of the variance in nitrogen removal results.

Comparing regression models for sediment and nitrogen removal (Figures 5.2 and 5.3) illustrates three important points regarding the pollutant removal function of riparian areas. First, the magnitude of removal efficacy for different pollutants can be very different. According to the models, 80% of sediment was removed at roughly 50 to 65 ft (15 to 20 m), but 80% removal of nitrogen may require buffer widths two to four times wider. Second, the variability of removal efficacy for various pollutants can be significantly different. The regression models for nitrogen removal show much more variation than the models for sediment removal. The wider variation is due, in part, to the greater complexity of the chemical and biological processes affecting nitrogen removal compared to the simpler physical processes affecting sediment removal. Hence, our ability to predict removal efficacy and our certainty about water quality protection can be very different for different pollutants. Third, for all pollutants the relationship between buffer width and removal efficacy follows a law of diminishing marginal returns. That is, the marginal (or incremental) amount of pollutant removal decreases as buffer width increases.

Zhang et al. (2010) also conducted meta-analysis for phosphorus and pesticides. Buffer width alone explained 35% of the variance in phosphorus removal results (Figure 5.5), and with the addition of vegetation as an independent variable, the model explained 47% of the variance. The buffer-width-only model predicts 89% phosphorus removal with a buffer 98 ft (30 m) wide.[7] Buffers with trees were found to remove more phosphorus than buffers completely or partially composed of grasses. The pesticide meta-analysis included 11 chemicals (norfluranzon, fluometuron, lindane, deisopropylatrazine, deethylatrazine, atrazine, permethrin, bromide, terbuthylazine, metolachlor, isoproturon). Buffer width alone explained 60% of the variance in pesticide removal results, and vegetation was found to be an insignificant predictor variable. The model predicts 92% pesticide removal with a buffer 65 ft (20 m) wide.

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[7] Because Zhang et al. (2010) assumed an asymptotic equation for their regression, the phosphorus model was invalid for efficacies greater than 89.5%.

Using data provided by Zhang et al. (2010), we calculated prediction intervals for their phosphorus model.[8] A prediction interval describes the precision of a model's predictions and the interval's width is a metric of uncertainty. For a buffer 33 ft (10 m) wide, the width of the 90% prediction interval is 64%. Therefore, while the model predicts that a 33 ft (10 m) buffer will provide an average removal efficacy of 70%, the actual removal efficacy obtained by another study on phosphorus removal (i.e., one which was not included in the meta-analysis) is likely to be somewhere between 38 and 100% (70% ± 32%). The low precision of the model's prediction interval is related to the unexplained variance that could be attributed to other variables not included in the regression, such as site slope and various soil properties.

Table 5.1. Descriptive information for functional relationships on sediment removal shown in Figure 5.2 and nitrogen removal shown in Figure 5.3. Reference numbers 1 through 4 correspond to numbers in Figures 5.2 and 5.3. In the equations, E_R is the removal efficacy, *width* is riparian buffer width, and *a, b, c, d,* and k_{50} are parameters determined through linear or nonlinear regression.

Pollutant	Meta-analysis	Form of equation	Model	*N****	R^2	*P* value
Sediment	(1) Liu et al. (2008)**	$E_R = a + b\ \ln(width)$	width	79	0.34	< 0.001
		$E_R = a + b \times width + c \times slope + d \times slope^2$	width & slope = 5% width & slope = 10% width & slope = 15%	79	0.43	< 0.0001
	(2) Yuan et al. (2009)	$E_R = a + b\ \ln(width)$	width	75	0.30	na*
			width & slope ≤ 5%	53	0.32	na
			width & slope > 5%	14	0.17	na
	(3) Zhang et al. (2010)**	$E_R = a\,(1 - e^{b \times width})$	width	81	0.37	< 0.0001
		$E_R = a\,(1 - e^{b \times width}) + c + d \times slope$	width & slope = 5% & grass or trees width & slope = 10% & grass or trees width & slope = 15% & grass or trees	81	0.65	< 0.001
	(4) Sweeney and Newbold (2014)	$E_R = \dfrac{width}{k_{50} + width}$	width	22	0.28	na
Nitrogen	(1) Mayer et al. (2007)	$E_R = a \times width^b$	width, surface water only	23	0.21	0.03
			width & herbaceous vegetation	32	0.21	0.009
			width & herbaceous/forest vegetation	11	0.39	0.04
	(2) Zhang et al. (2010)	$E_R = a\,(1 - e^{b \times width})$	width	61	0.44	< 0.0001
		$E_R = a\,(1 - e^{b \times width}) + c$	width & trees vegetation width & grass or mixed grass/trees veg.	61	0.49	< 0.001
	(3) Sweeney and Newbold (2014)†	$E_R = 100\,(1 - e^{b \times width/q})$	width & q set to 58‡ width & q set to 115	30	0.37	na*

* Signifies that *P* value was not reported in cited study.

** Nearly all the studies included by Liu et al. (2008) and Zhang et al. (2010) in their meta-analyses were the same studies.

*** *N* signifies number of studies in meta-analysis.

† Studies for subsurface nitrate removal only.

‡ q represents subsurface water flux. 58 and 115 L/m/day are median and mean subsurface water fluxes for studies collected by Sweeney and Newbold (2014).

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[8] A prediction interval is a range that is likely to contain the dependent response value (Y_i) for a new observation (X_i). A prediction interval assumes that the new observation is taken from the same population used to create the regression equation.

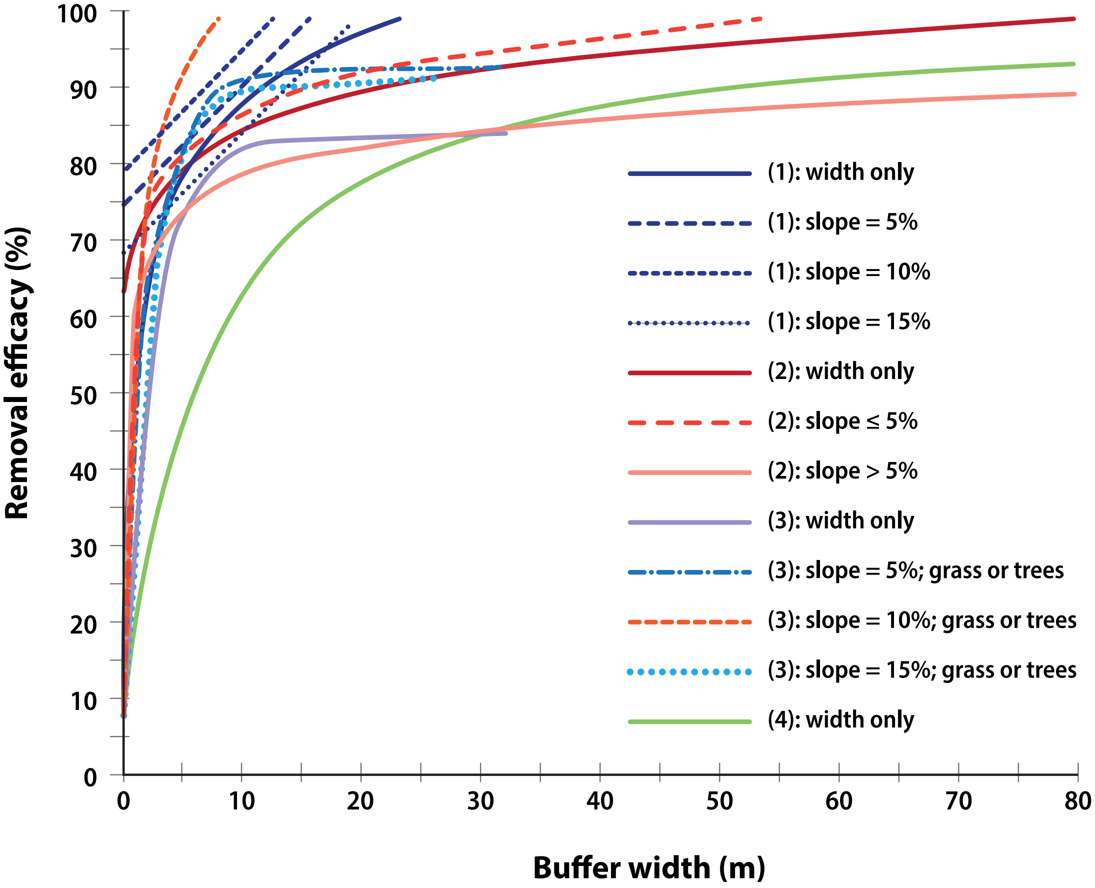

Figure 5.2. Relationships between sediment removal efficacy and riparian buffer width developed by four separate meta-analyses. Numbers in parentheses refer to different meta-analyses. See descriptive information in Table 5.1.

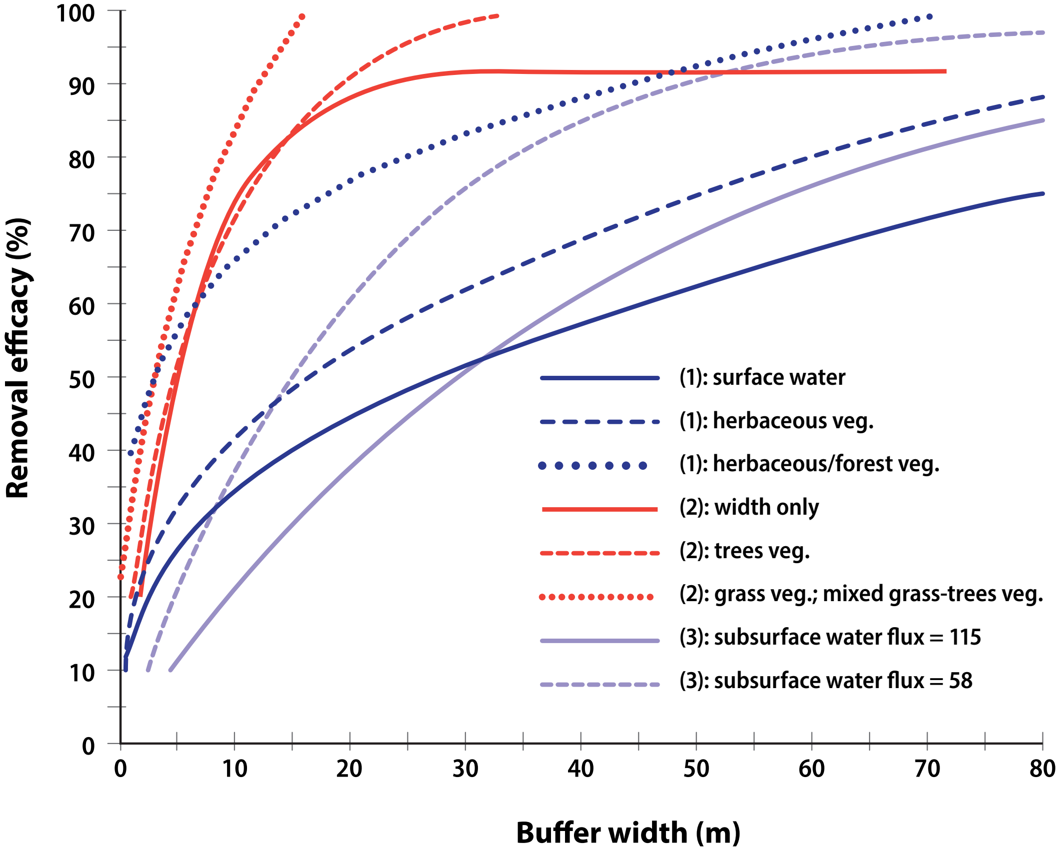

Figure 5.3. Relationships between nitrogen removal efficacy and riparian buffer width developed by three separate meta-analyses. Numbers in parentheses refer to different meta-analyses. See descriptive information in Table 5.1.

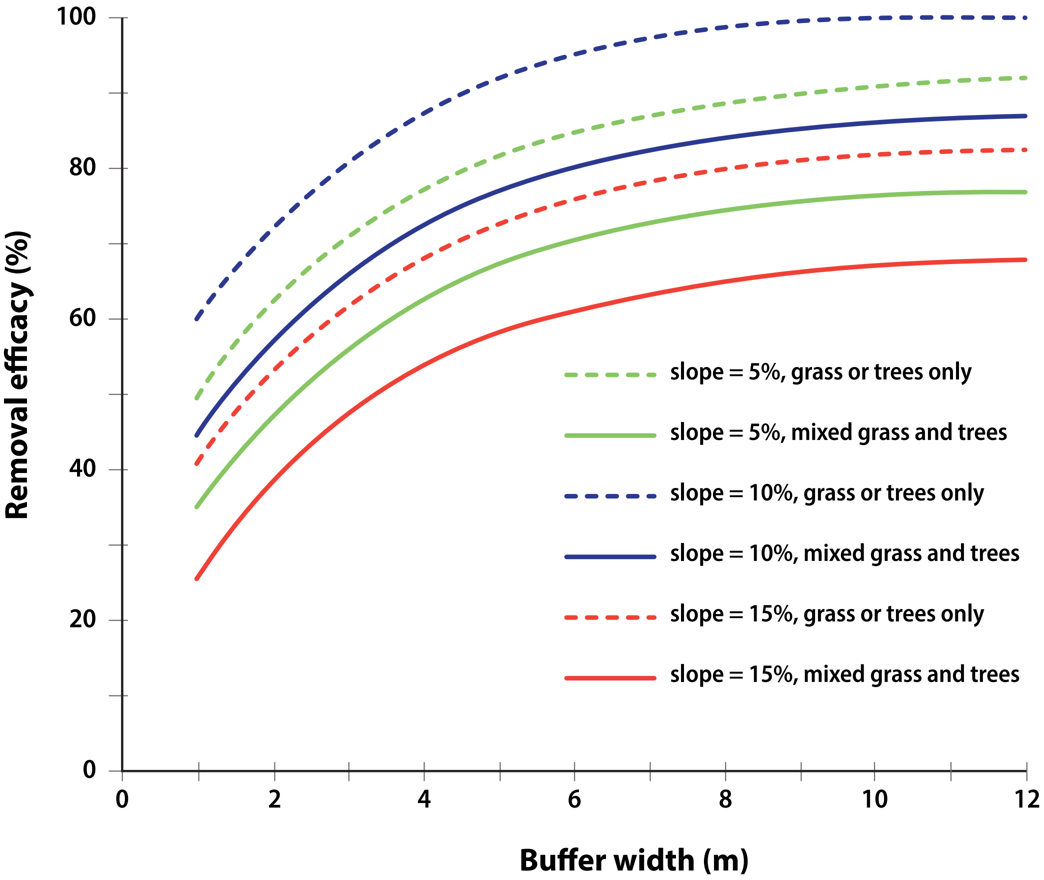

Figure 5.4. A result from meta-analysis of Zhang et al. (2010). Sediment removal efficacy as a function of buffer width, site slope, and buffer vegetation ($N = 81$, $R^2 = 0.65$, $P < 0.001$).

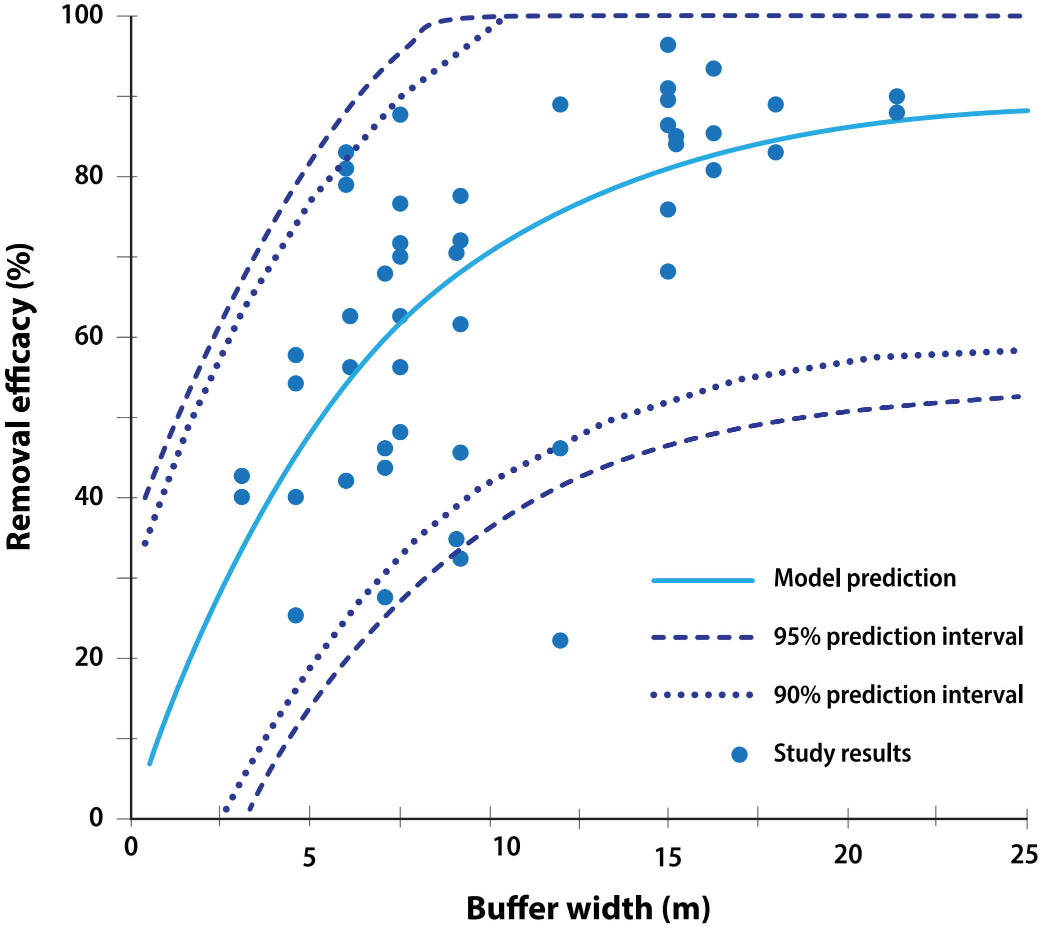

Figure 5.5. Meta-analysis of Zhang et al. (2010). Results of nonlinear regression for phosphorus removal efficacy as a function of buffer width ($N = 52$, $R^2 = 0.35$, $P < 0.0001$) with 90% and 95% prediction intervals.

5.6. Reanalysis of Meta-Analyses

A major assumption made by each of the meta-analyses was the type of equation that would result in the best fit to their data. Zhang et al. (2010), for instance, chose to use a nonlinear asymptotic equation of the form:

$$E_R = a\,(1 - e^{b \times width}) \qquad (2)$$

where E_R is removal efficacy, ***width*** refers to buffer width, and ***a*** and ***b*** are coefficients determined through nonlinear regression. In equation (2), *a* determines the maximum removal efficacy possible (i.e., it is an asymptote). Zhang et al. (2010) conducted their regression such that *a* could be less than 100%. For sediment, pesticide, nitrogen, and phosphorus pollutants, *a* equaled 91, 93, 92, and 89.5 percent, respectively. Hence, we could not determine buffer widths needed for removal efficacies greater than these values of *a* in their models. This limitation was particularly problematic for pesticides and phosphorus because Zhang et al. (2010) was the only meta-analysis for those pollutants.

To determine buffer widths needed for higher removal efficacies (90, 95, 99%), we reanalyzed Zhang et al.'s data with a nonlinear equation of the form:

$$E_R = a + b\ \ln(width) \qquad (3)$$

This form of model is the same used by Liu et al. (2008) and Yuan et al. (2009).[1] We performed linear regression in R (RCT 2013) using the lm function, but first we established that we could obtain the same regression results (same values for ***a*** and ***b***) as Zhang et al. (2010) using equation (2). This confirmed that the data we downloaded from the internet were the same data used by Zhang et al. (2010). We then redid the regression using equation (3). With the exception of sediment, the resulting models fit the data as well or better than the original models of Zhang et al. (2010) (Table 5.2). Graphical comparisons of our results and those of Zhang et al. (2010) are presented in Figure 5.6.

Table 5.2. Comparison of R^2 values for Zhang et al.'s (2010) model and our reanalysis using equation (3). *N* is the number of studies included in the meta-analysis.

Pollutant	*N*	R^2	
		Zhang et al. (2010)	Reanalysis
Sediment	81	0.37	0.31
Nitrogen	61	0.44	0.44
Phosphorus	52	0.35	0.39
Pesticides	49	0.60	0.61

Juvenile fish find shelter in a root wad/Terra Hegy, WDFW

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[1] Xuyang Zhang, the lead author of Zhang et al. (2010), also recommended that we use equation (3) because equation (2) cannot model 100% removal efficacy (X. Zhang, California Department of Pesticide Regulation, personal communication, May 14, 2019).

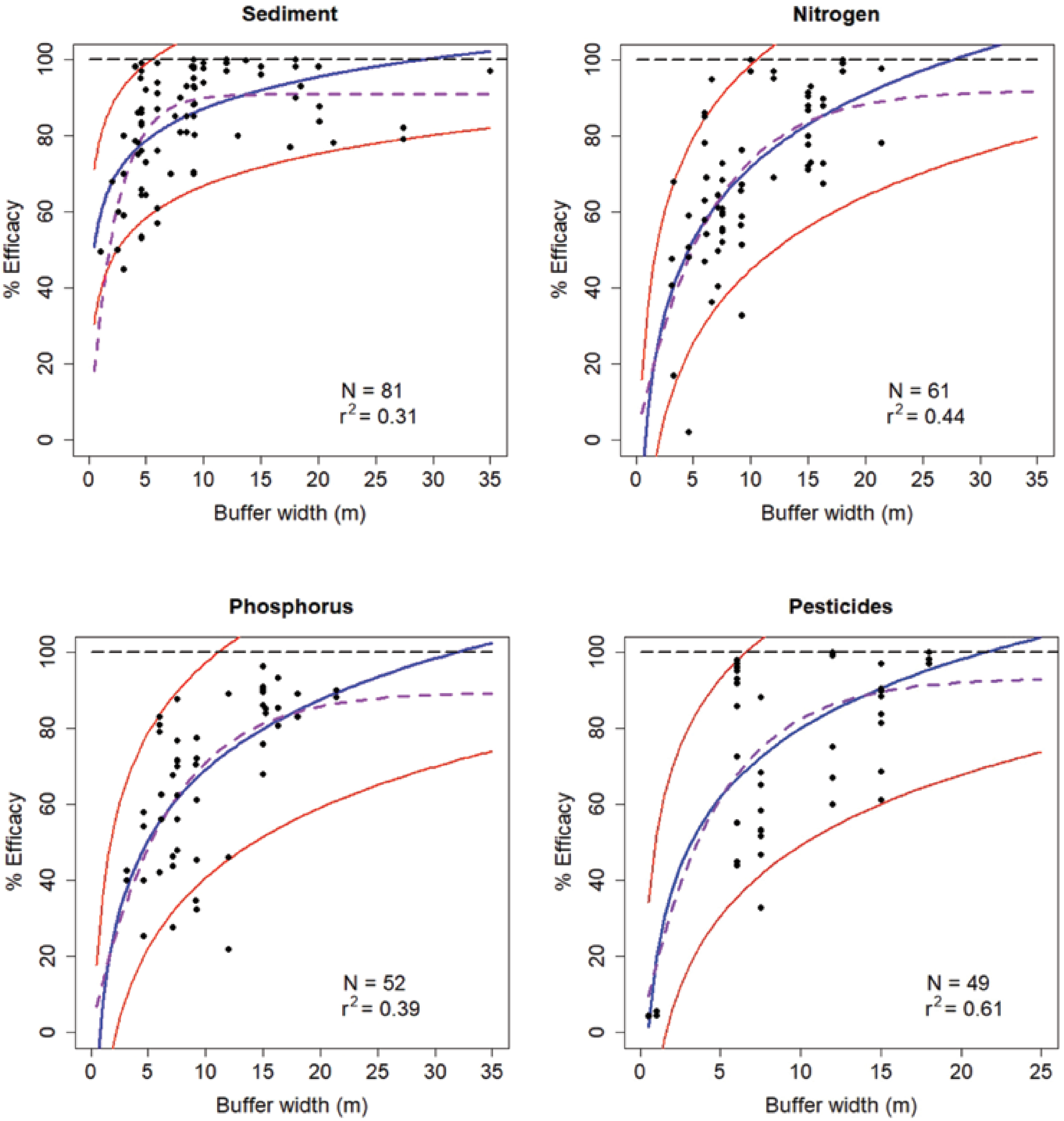

Figure 5.6. Relationship between removal efficacy and buffer width. Data from meta-analysis of Zhang et al. (2010) but reanalyzed using different form of nonlinear model. Line colors: blue: best fit model, red: 90% prediction interval, purple dashed: original model of Zhang et al. (2010), black dashed: 100% removal efficacy

We also reanalyzed two other meta-analyses – Mayer et al. (2007), and Sweeney and Newbold (2014) – for two reasons. The first reason was consistency across meta-analyses. The meta-analyses gave different answers for the relationship between removal efficacy and buffer width. For example, Zhang et al. (2010) predicted that buffer widths of 80 ft (24 m) are necessary for 90% removal efficacy of nitrogen, while Mayer et al. (2007) predicted buffer widths of 436 ft (133 m) for the same efficacy. Part of this discrepancy may be due to the type of equations adopted for their nonlinear or linear regressions, so we redid both meta-analyses using the same equation. Second, as noted earlier, a major assumption made by each of the meta-analyses was the type of equation that would result in the best fit to their data. We tested that assumption, and found that equation (3) often resulted in a better fit.

Mayer at al. (2007) chose to use a power function of the form:[2]

$$E_R = a\ width^b \qquad (4)$$

where *a* and *b* are coefficients determined through linear regression. We reanalyzed their data using equation (3). The resulting models fit the data as well or better than the original models of Mayer et al. (2007) (Table 5.3, Figure 5.7). For subsurface flow, the relationship between removal efficacy and buffer width was statistically insignificant ($P = 0.30$) and R^2 was only 0.02 for both their model and ours. We also note that both models produce somewhat absurd results – 60% of subsurface nitrogen can be removed with buffers only 2 ft wide but 90% removal requires buffers over 1200 ft wide. Hence, we will use neither model in further discussions of sub-surface nitrogen removal. For surface flow, the reanalysis resulted in a slightly better fit to the data – R^2 improved from 0.21 to 0.28.

We did not reanalyze the meta-analysis that Sweeny and Newbold (2014) did on studies of nitrogen removal from groundwater. While Sweeney and Newbold (2014) found a significant statistical relationship between nitrogen removal from subsurface flow and buffer width, other studies suggest the buffer width has little or no effect on nitrogen removal from subsurface flow (Hruby 2013, pp. 10-11).

Table 5.3. Comparison of R^2 values for Mayer et al.'s (2007) model and our reanalysis using equation (3). *N* is the number of studies included in the meta-analysis.

		Mayer et al. (2007)		Reanalysis	
Model	*N*	R^2	*P*	R^2	*P*
nitrogen, surface	23	0.21	0.03	0.28	0.009
nitrogen, subsurface	65	0.02	0.30	0.02	0.30

For sediment removal, Sweeney and Newbold (2014) chose an equation of the form:

$$E_R = \frac{width}{k_{50} + width} \qquad (5)$$

where k_{50} equals the buffer width expected to remove 50% of sediments and it is determined through nonlinear regression. We reanalyzed their data using equation (3). The resulting model fits the data better than the original model of Sweeney and Newbold (2007). Their R^2 was 0.28 and the new R^2 is 0.33 (Figure 5.7).

Table 5.4 shows the buffer width needed to achieve desired levels of pollutant removal as predicted by meta-analyses and our reanalyses of meta-analyses. We combined predictions from multiple meta-analyses using a weighted average with the weights being the products of *N* and R^2. Note that values in Table 5.4 are predicted averages for removal efficacy. The actual efficacy realized by implementing a particular buffer width could be much better or worse. The amount of uncertainty in the realized removal efficacy is depicted by prediction intervals in Figures 5.6 and 5.7. The

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[2] The equation $E_R = a\ width^b$ can be transformed to the equation $\ln(E_R) = a' + b\ \ln(width)$, which is similar in structure to equation (3) and results in a similar fit to the data. Equation (3) was also the form used by Mayer et al. (2006).

Riparian areas capture pollutants/Marlin Greene, One Earth Images

reanalyzed meta-analysis of Zhang et al. (2010), for example, predicts that 90% of sediment can be removed with a 42 ft (13 m) wide buffer, however, according to the prediction interval, the buffer width needed for 90% removal could be as narrow as 8 ft (2.4 m) or as wide as 225 ft (69 m). Better models that include other independent variables, such as slope and vegetation (e.g., Zhang et al. 2010), will reduce this uncertainty, but there is a trade-off. The application of more complicated models leads to more complicated management. If, for example, managers adopted a model that includes vegetation type or vegetation structure, then, in addition to buffer width, they must also manage the condition of vegetation within the buffer.

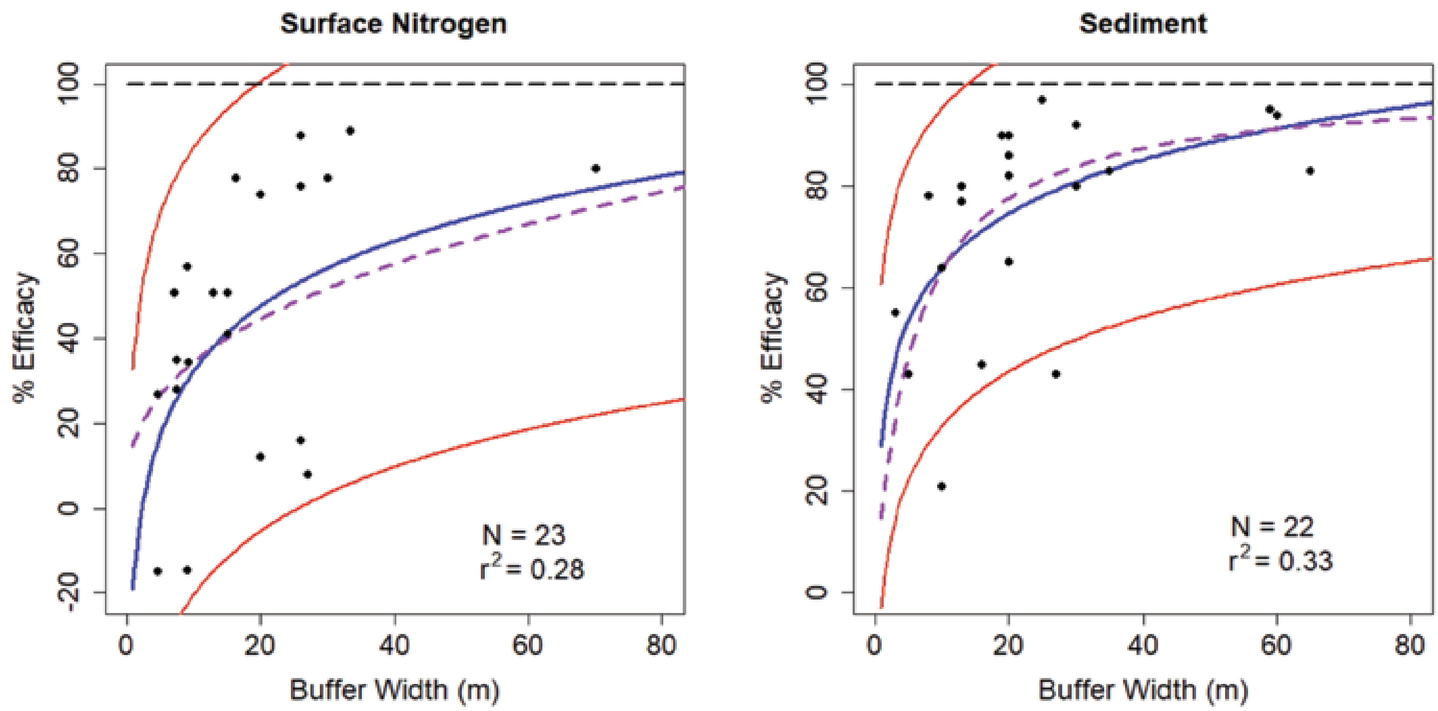

Figure 5.7. Relationship between removal efficacy and buffer width. Study results from meta-analyses reanalyzed using different form of equation in a linear regression. Left: nitrogen study results from Mayer et al. (2007). Right: sediment study results from Sweeney and Newbold (2014). Line colors: blue: best fit model, red: 90% prediction interval, purple dashed: original models from meta-analyses, black dashed: 100% removal efficacy.

Table 5.4. Buffer width in feet needed to achieve desired levels of pollutant removal from run-off. Buffer width calculated with model and/or data from the cited literature. Models had only one predictor variable, which was buffer width. Mean buffer width for sediment and nitrogen removal are weighted averages of widths determined by each meta-analysis. Weights were the product of *N* and R^2.

			Average Removal Efficacy (%)	**80**	**90**	**95**	**99**
Pollutant	**Meta-analysis**	***N***	**R^2**	**Buffer Width (ft)**			
Sediment	Liu et al. (2008)*	79	0.34	18	39	56	76
	Yuan et al. (2009)	75	0.30	18	74	149	261
	Zhang et al. (2010)*, *reanalyzed*	81	0.31	18	42	64	89
	Sweeney and Newbold (2014), *reanalyzed*	22	0.33	94	181	251	326
			Weighted mean	**25**	**62**	**101**	**153**
Nitrogen, surface	Mayer et al. (2007), *reanalyzed*	23	0.28	283	444	556	666
	Zhang et al. (2010), *reanalyzed*	61	0.44	44	63	76	88
			Weighted mean	**90**	**137**	**169**	**200**
Nitrogen, subsurface	Sweeney and Newbold (2014) ‡	30	0.37	112	161	210	322
Phosphorus	Zhang et al. (2010), *reanalyzed*	52	0.39	50	72	87	101
Pesticides	Zhang et al. (2010), *reanalyzed*	49	0.61	33	48	59	68

* Nearly all the studies included by Liu et al. (2008) and Zhang et al. (2010) were the same studies.

‡ The parameter q, which represents subsurface water flux, was set to 58, which is the median value for 30 studies analyzed by Sweeney and Newbold (2014).

5.7. Conclusions

There is a consensus in the scientific literature that riparian areas reduce the flow of pollutants to aquatic ecosystems, and that pollutant removal functions are contingent on complex interactions between hydrology, soil, and vegetation, and are influenced by many other factors as well. Because hydrology, soil, vegetation and other factors exhibit high spatial and temporal variability, the efficacy of pollutant removal functions provided by riparian areas differs substantially among sites, as well as within sites. Determining the potential for riparian areas to protect water quality is further complicated by the disparate behavior of different contaminant types in the environment. While space limitations required us to simplify this review of the scientific literature regarding pollutant removal functions of riparian areas, we provided the basic information needed for understanding these important ecological functions.

Our review of the literature presents a substantial body of scientific research with which to develop strategies, plans, or policies regarding the pollutant removal functions of riparian areas. The central problem faced by resource managers is determining the adequate riparian buffer width, composition, and structure to protect water quality with high degrees of efficacy, efficiency, and certainty. Riparian function is often simplistically characterized by buffer width and vegetation type, but these parameters only partially explain the effects that riparian areas have on pollutant removal. The current conservation paradigm remains strongly tied to the premise that wider riparian areas are the only way to increase pollutant removal efficacy, but this is not always the case; other site-specific factors, such as conditions within the buffer, also exert a significant influence on removal efficacy. Therefore, consideration of the entire system (i.e., riparian area, uplands, vegetation, soils, ground and surface water pathways, topography, type of pollutant, etc.) is essential to developing cost-effective pollutant removal. In theory, buffer width and vegetation could be optimized for site conditions such that water quality is protected while minimizing economic costs for landowners.

Despite the many scientific uncertainties, management decisions must be made. Management decisions regarding pollutant removal by riparian buffers should be informed by science, but determining the "right" buffer width for pollutant removal cannot be purely scientific. Determining the "right" buffer width begins with choosing a desired removal efficacy, and that choice is normative. That is, the desired removal efficacy is a social choice influenced by cultural values, economic costs, and risk tolerance. Choosing a desired removal efficacy and determining the "right" buffer requires: 1) factual information regarding the anticipated impacts or outcomes of policy options (i.e., science); 2) an understanding of stakeholders' priorities and preferences (i.e., values); and 3) a process for using science and values to explore tradeoffs amongst policy options (Wilhere and Quinn 2018).

5.8. Literature Cited[3]

Ahmad, M., A.U. Rajapaksha, J.E. Lim, M. Zhang, N. Bolan, D. Mohan, M. Vithanage, S.S. Lee, and Y.S. Ok. 2014. Biochar as a sorbent for contaminant management in soil and water: a review. Chemosphere 99:19-33. (i)

Allaire, S.E., C. Sylvain, S.F. Lange, G. Thériault, and P. Lafrance. 2015. Potential efficiency of riparian vegetated buffer strips in intercepting soluble compounds in the presence of subsurface preferential flows. PLoS ONE 10: e0131840. (i)

Alvarez, M.D., and L.A. Fuiman. 2005. Environmental levels of atrazine and its degradation products impair survival skills and growth of Red Drum larvae. Aquatic Toxicology 74:229-241. (i)

Anbumozhi, V., J. Radhakrishnan, and E. Yamaji. 2005. Impact of riparian buffer zones on water quality and associated management considerations. Ecological Engineering 24:517-523. (i)

Angier, J.T., G.W. McCarty, C.P. Rice, and K. Bialek. 2002. Influence of a riparian wetland on nitrate and herbicides exported from an agricultural field. Journal of Agricultural and Food Chemistry 50:4424-4429. (i)

Au, D.W.T., C.A. Pollino, R.S.S. Wu, P.K.S. Shin, S.T.F. Lau, and J.Y.M. Tang. 2004. Chronic effects of suspended solids on gill structure, osmoregulation, growth, and triiodothyronine in juvenile Green Grouper *Epinephelus coioides*. Marine Ecology Progress Series 266:255-264. (i)

Backer, L.C., J.H. Landsberg, M. Miller, K. Keel, and T.K. Taylor. 2011. Canine cyanotoxin poisonings in the United States (1920s-2012): review of suspected and confirmed cases from three data sources. Toxins 5:1597-1628. (i)

Baldwin, D.H., J.F. Sandahl, J.S. Labenia, and N.L. Scholz. 2003. Sublethal effects of copper on Coho Salmon: impacts on non-overlapping receptor pathways in the peripheral olfactory nervous system.

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[3] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Environmental Toxicology and Chemistry 22:2266-2274. (i)

Barling, R.D., and I.D. Moore. 1994. Role of buffer strips in management of waterway pollution: a review. Environmental Management 18:543-558. (i)

BLM (Bureau of Land Management). 2003. Riparian area management: riparian-wetland soils. Technical Reference 1737-19. Denver. (viii)

Boening, D.W. 2000. Ecological effects, transport, and fate of mercury: a general review. Chemosphere 40:1335-1351. (i)

Bohn, C.C., and J.C. Buckhouse. 1985. Some responses of riparian soils to grazing management in northeastern Oregon. Journal of Range Management 38:378-381. (i)

Bolan, N., A. Kunhikrishnan, R. Thangarajan, J. Kumpiene, J. Park, T. Makino, M.B. Kirkham, and K. Scheckel. 2014. Remediation of heavy metal(loid)s contaminated soils—to mobilize or to immobilize? Journal of Hazardous Materials 266:141-166. (i)

Boonchan, S., M.L. Britz, and G.A. Stanley. 2000. Degradation and mineralization of high-molecular-weight polycyclic aromatic hydrocarbons by defined fungal-bacterial cocultures. Applied and Environmental Microbiology 66:1007-1019. (i)

Bourret, M.M., J.E. Brummer, and W.C. Leininger. 2009. Establishment and growth of two willow species in a riparian zone impacted by mine tailings. Journal of Environmental Quality 38:693-701. (i)

Bradford, S.A., and S. Torkzaban. 2008. Colloid transport and retention in unsaturated porous media: a review of interface-, collector-, and pore-scale processes and models. Vadose Zone Journal 7:667-681. (i)

Buffler, S., C. Johnson, J. Nicholson, and N. Mesner. 2005. Synthesis of design guidelines and experimental data for water quality function in agricultural landscapes in the Intermountain West. National Agroforestry Center, Lincoln, Nebraska. (viii)

Burt, T.P., L.S. Matchett, W.T. Goulding, C.P. Webster, and N.E. Haycock. 1999. Denitrification in riparian buffer zones: the role of floodplain hydrology. Hydrologic Processes 13:1451-1463. (i)

Canet, R., J.G. Brinsting, D.G. Malcolm, J.M. Lopez-Real, and A.J. Beck. 2001. Biodegradation of polycyclic aromatic hydrocarbons (PAHs) by native microflora and combinations of white-rot fungi in a coal-tar contaminated soil. Bioresource Technology 76:113-117. (i)

Castelle, A.J., A.W. Johnson, and C. Conolly. 1994. Wetland and stream buffer size requirements—a review. Journal of Environmental Quality 23:878-882. (i)

Cedergreen, N. 2014. Quantifying synergy: a systematic review of mixture toxicity studies within environmental toxicology. PLoS ONE 9:e96580. (i)

Cey E.E., D.L. Rudolph, R. Aravena, and G. Parkin. 1999. Role of the riparian zone in controlling the distribution and fate of agricultural nitrogen near a small stream in southern Ontario. Journal of Contaminant Hydrology 37:45–67. (i)

Chaubey, I., D.R. Edwards, T.C. Daniel, P.A. Moore, and D.J. Nichols. 1995. Effectiveness of vegetative filter strips in controlling losses of surface-applied poultry litter constituents. Transactions of the American Society of Agricultural Engineers 38:1687-1692. (i)

Cheng, Z.Q., E.L. McCoy, and P.S. Grewal. 2014. Water, sediment, and nutrient runoff from urban lawns established on disturbed subsoil or topsoil and managed with inorganic or organic fertilizers. Urban Ecosystems 17:277-289. (i)

Clements, W.H., D.S. Cherry, and J. Cairns, Jr. 1990. Macroinvertebrate community responses to copper in laboratory and field experimental streams. Archives of Environmental Contaminants and Toxicology 19:361-365. (i)

Clinton, B.D. 2011. Stream water responses to timber harvest: riparian buffer width effectiveness. Forest Ecology and Management 261:979-988. (i)

Colborn, T., F.S.V. Saal, and A.M. Soto. 1993. Developmental effects of endocrine-disrupting chemicals in wildlife and humans. Environmental Health Perspectives 101:378-384. (i)

Collins, R., A. Donnison, C. Ross, and M. McLeod. 2004. Attenuation of effluent-derived faecal microbes in grass buffer strips. New Zealand Journal of Agricultural Research 47:565-574. (i)

Cooper, J.R., and J.W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. Soil Science Society of American Journal 51:1600-1604. (i)

Cors, M., and B. Tychon. 2007. Grassed buffer strips as nitrate diffuse pollution remediation tools: management impact on the denitrification enzyme activity. Water Science and Technology 55:25-30. (i)

Correll, D.L. 1997. Buffer zones and water quality protection: general principles. Pages 7-20 *in* N. Hancock, T. Burt, K. Goulding, and G. Pinay, editors. Buffer zones: their processes and potential in water protection. Quest Environmental, Hertfordshire, United Kingdom. (viii)

Covaci, A., S. Harrad, M.A.E. Abdallah, N. Ali, R.J. Law, D. Herzke, and C.A. de Wit. 2011. Novel brominated flame retardants: a review of their analysis, environmental fate and behavior. Environment International 37:532-556. (i)

Coyne, M.S., R.A. Gilfillen, R.W. Rhodes, and R.L. Blevins. 1995. Soil and fecal-coliform trapping by grass filter strips during simulated rain. Journal of Soil and Water Conservation 50:405-408. (i)

Coyne, M.S., R.A. Gilfillen, A. Villalba, Z. Zhang, R. Rhodes, L. Dunn, and R.L. Blevins. 1998. Fecal bacteria trapping by grass filter strips during simulated rain. Journal of Soil and Water Conservation 53:140-145. (i)

Daniels, R.B., and J.W. Gilliam. 1996. Sediment and chemical load reduction by grass and riparian filters. Soil Science Society of America Journal 60:246-251. (i)

De Snoo, G.R., and P.J. De Witt. 1998. Buffer zones for reducing pesticide drift to ditches and risks to aquatic organisms. Ecotoxicology and Environmental Safety 41:112-118. (i)

Delle Site, A. 2001. Factors affecting sorption of organic compounds in natural sorbent/water systems and sorption coefficients for selected pollutants—a review. Journal of Physical and Chemical Reference Data 30:187-439. (i)

Dillaha, T.A., R.B. Reneau, S. Mostaghimi, and D. Lee. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. Transactions of American Society of Agricultural Engineers 32:513-519. (i)

Dorioz, J.M., D. Wang, J. Poulenard, and D. Trévisan. 2006. The effect of grass buffer strips on phosphorus dynamics—a critical review and synthesis as a basis for application in agricultural landscapes in France. Agriculture, Ecosystems and Environment 117:4-21 (i)

Dosskey, M.G. 2001. Toward quantifying water pollution abatement in response to installing buffers on crop land. Environmental Management 28:577-598. (i)

Dosskey, M.G., M.J. Helmers, D.E. Eisenhauer, T.G. Franti, and K.D. Hoagland. 2002. Assessment of concentrated flow through riparian buffers. Journal of Soil and Water Resources Association 57:336-343. (i)

Dosskey, M.G., P. Vidon, N.P. Gurwick, C.J. Allan, T.P. Duval, and R. Lowrance. 2010. The role of riparian vegetation in protecting and improving chemical water quality in streams. Journal of the American Water Resources Association 46:261-277. (i)

Doyle, R.C., D.C. Wolf, and D.F. Bezdicek. 1975. Effectiveness of forest buffer strips in improving the water quality of manure polluted runoff. Pages 299-302 *in* Managing livestock wastes. Proceedings of 3rd international symposium on livestock wastes. Food and Agricultural Organization of the United Nations, Rome. (viii)

Du Laing, G., J. Rinklebe, B. Vandecasteele, E. Meers, and F.M.G. Tack. 2009. Trace metal behavior in estuarine and riverine floodplain soils and sediments: a review. Science of the Total Environment 407:3972-3985. (i)

Ecology (Washington State Department of Ecology). 2007. Mission Creek watershed DDT total maximum daily load: water quality improvement report. Publication number 07-10-046. Olympia, Washington. (viii)

Entry, J.A., R.K. Hubbard, J.E. Thies, and J.J. Fuhrmann. 2000. The influence of vegetation in riparian filterstrips on coliform bacteria: II. Survival in soils. Journal of Environmental Quality 29:1215-1224. (i)

Euliss, K., C. Ho, A.P. Schwab, S. Rock, and M.K. Banks. 2008. Greenhouse and field assessment of phytoremediation for petroleum contaminants in a riparian zone. Bioresource Technology 99:1961-1971. (i)

Fajardo, J.J., J.W. Bauder, and S.D. Cash. 2001. Managing nitrate and bacteria in runoff from livestock confinement areas with vegetative filter strips. Journal of Soil and Water Conservation 56:185-191. (i)

Feng, J.C., D.G. Thompson, and P.E. Reynolds. 1990. Fate of glyphosate in a Canadian Forest Watershed. 1. Aquatic residues and off-target deposit assessment. Journal of Agricultural and Food Chemistry 38:1110-1118. (i)

Fennesy, M.S., and J.K. Cronk. 1997. The effectiveness and restoration potential of riparian ecotones for the management of nonpoint source pollution, particularly nitrate. Critical Reviews in Environmental Science and Technology 27:285-317. (i)

Fisher, R.F., and D. Binkley. 2000. Ecology and management of forest soils. Wiley, New York. (i)

Fischer, R.A., and J.C. Fischenich. 2000. Design recommendations for riparian corridors and vegetative buffer strips. U.S. Army Engineer Research and Development Center, Vicksburg, Mississippi. (viii)

Fleeger, J.W., K.R. Carman, and R.M. Nisbet. 2003. Indirect effects of contaminants in aquatic ecosystems. Science of the Total Environment 317:207-233. (i)

Fox, G.A., D.M. Heeren, R.B. Miller, A.R. Mittelstet, and D.E. Storm. 2011a. Flow and transport experiments for a streambank seep originating from a preferential flow pathway. Journal of Hydrology 403:360-366. (i)

Fox, G.A., E.M. Matlock, J.A. Guzman, D. Sahoo, and K.B. Stunkel. 2011b. *Escherichia coli* load reduction from runoff by vegetative filter strips: a laboratory-scale study. Journal of Environmental Quality 40:980-988. (i)

Fuchs, J.W., G.A. Fox, D.E. Storm, C.J. Penn, and G.O. Brown. 2009. Subsurface transport of phosphorus in riparian floodplains: influence of preferential flow paths. Journal of Environmental Quality 38:473-484. (i)

Garcia-Muñoz, E., F. Guerrero, and G. Parra. 2011. Larval escape behavior in anuran amphibians as a wetland rapid pollution biomarker. Marine and Freshwater Behaviour and Physiology 44:109-123. (i)

Ghosh, M., and S.P. Singh. 2005. A review of phytoremediation of heavy metals and utilization of its byproducts. Applied Ecology and Environmental Research 3:1-18. (i)

Gonod, L.V., F. Martin-Laurent, and C. Chenu. 2006. 2,4-D impact on bacterial communities, and the activity and genetic potential of 2,4-D degrading communities in soil. FEMS Microbiology Ecology 58:529-537. (i)

Gordon, N.D., T.A. McMahon, B.L. Finlayson, C.J. Gippel, and R.J. Nathan. 2004. Stream hydrology: an introduction for ecologists. Wiley, West Sussex, England. (i)

Grathwohl, P. 1990. Influence of organic matter from soils and sediments from various origins on the sorption of some chlorinated aliphatic hydrocarbons: Implications on K_{OC} correlations. Environmental Science & Technology 24:1687-1693. (i)

Groffman, P.M., E.A. Axelrod, J.L. Lemonyon, and W.M. Sullivan. 1991. Dentrification in grass and forest vegetated filter strips. Journal of Environmental Quality 20:671-674. (i)

Gu, C., W. Anderson, and F. Maggi. 2012. Riparian biogeochemical hot moments induced by stream fluctuation. Water Resources Research 48:W09546. (i)

Guan, K., S.E. Thompson, C.J. Harman, N.B. Basu, P.S.C. Rao, M. Sivapalan, A.I. Packman, and P.K. Kalita. 2011. Spatiotemporal scaling of hydrological and agrochemical export dynamics in a tile-drained midwestern watershed. Water Resources Research 47:W00J02. (i)

Guber, A.K., A.M. Yakirevich, A.M. Sadeghi, Y.A. Pachepsky, and D.R. Shelton. 2009. Uncertainty evaluation of coliform bacteria removal from vegetative filter strip under overland flow condition. Journal of Environmental Quality 38:1636-1644. (i)

Haritash, A.K., and C.P. Kaushik. 2001. Biodegradation aspects of polycyclic aromatic hydrocarbons (PAHs): a review. Journal of Hazardous Materials 169:1-15. (i)

Hawes, E., and M. Smith. 2004. Riparian buffer zones: functions and recommended widths. Yale School of Forestry and Environmental Studies, New Haven, Connecticut. (viii)

Hayashi, M., and D.O. Rosenberry. 2002. Effects of ground water exchange on the hydrology and ecology of surface water. Ground Water 40:309–316. (i)

Hayes, M.H.B. 1970. Adsorption of triazine herbicides on soil organic matter, including a short review on soil organic matter chemistry. Residue Reviews 32:137-174. (i)

Hayes, T., K. Haston, M. Tsui, A. Hoang, C. Haeffele, and A. Vonk. 2003. Atrazine-induced hermaphroditism at 0.1 ppb in American Leopard Frogs *Rana pipiens*: laboratory and field evidence. Environmental Health Perspectives 111:568-575. (i)

Hecht, S.A., D.H. Baldwin, C.A. Mebane, T. Hawkes, S.J. Gross, and N.L. Scholz. 2007. An overview of sensory effects on juvenile salmonids exposed to dissolved copper: applying a benchmark concentration approach to evaluate sublethal neurobehavioral toxicity. Tech Memo NMFS-NWFSC-83. National Marine Fisheries Service, Seattle. (viii)

Hedin, L.O., J.C. von Fischer, N.E. Ostrom, B.P. Kennedy, M.G. Brown, and G.P. Robertson. 1998. Thermodynamic constraints on nitrogen transformations and other biogeochemical processes at soil-stream interfaces. Ecology 79:684-703. (i)

Hickey, M.B., and B. Doran. 2004. A review of efficiency of buffer strips for the maintenance and enhancement of riparian ecosystems. Water Quality Research Journal of Canada 39:311-317. (i)

Hoffmann, C.C., C. Kjaergaard, J. Uusi-Kämppä, H.C.B. Hansen, and B. Kronvang. 2009. Phosphorus retention in riparian buffers: review of their efficiency. Journal of Environmental Quality 38:1942-1955. (i)

Hill, A.R. 1996. Nitrate removal in stream riparian zones. Journal of Environmental Quality 25:743-755. (i)

Hruby, T. 2013. Update on wetland buffers: the state of the science, final report. Washington State Department of Ecolgy, Olympia, Washington. (viii)

Hua, J., and R. Relyea. 2014. Chemical cocktails in aquatic systems: pesticide effects on the response and recovery of >20 animal taxa. Environmental Pollution 189(June):18-26. (i)

Huggenberger, P., E. Hoehn, R. Beschta, and W. Woessner. 1998. Abiotic aspects of channels and floodplains in riparian ecology. Freshwater Biology 40:407-425. (i)

Ingersoll, C.G., and R.W. Winner. 1982. Effect on *Daphnia pulex* (De Geer) of daily pulse exposures to copper or cadmium. Environmental Toxicology and Chemistry 1:321-327. (i)

Jarvie, H.P., A.N. Sharpley, B. Spears, A.R. Buda, L. May, and P.J.A. Kleinman. 2013. Water quality remediation faces unprecedented challenges from "legacy phosphorus". Environmental Science & Technology 47:8997–8998. (i)

Jaynes, D.B., and T.M. Isenhart. 2014. Reconnecting tile drainage to riparian buffer hydrology for enhanced nitrate removal. Journal of Environmental Quality 43:631-638. (i)

Jiang, X., J. Morgan, and M.P. Doyle. 2002. Fate of *Escherichia coli* O157:H7 in manure-amended soil. Applied and Environmental Microbiology 68:2605-2609. (i)

Johnson, C.W., and S. Buffler. 2008. Riparian buffer design guidelines for water quality and wildlife habitat functions on agricultural landscapes in the Intermountain West. RMRS-GTR-203. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado. (viii)

Jones, G.K. 2009. Bioremediation of contaminated riparian zones using mycorrhizal fungi—an exploration of the feasibility of restoration through mycoremediation. Master's thesis. The Evergreen State College, Olympia, Washington. (i)

Kalbus, E., C. Schmidt, J.W. Molson, F. Reinstorf, and M. Schirmer. 2009. Influence of aquifer and streambed heterogeneity on the distribution of groundwater discharge. Hydrology and Earth System Sciences 13:69-77. (i)

Karthikeyan, R., L.C. Davis, L.E. Erickson, K. Al-Khatib, P.A. Kulakow, P.L. Barnes, S.L. Hutchinson, and A.A. Nurzhanova. 2004. Potential for plant-based remediation of pesticide-contaminated soil and water using nontarget plants such as trees, shrubs, and grasses. Critical Reviews in Plant Sciences 23:91-101. (i)

Khan, S.U. 1972. Adsorption of pesticide by humic substances—a review. Environmental Letters 3:1-12. (i)

Kopinke, F.D., A. Georgi, and K. Mackenzie. 2001. Sorption of pyrene to dissolved humic substances and related model polymers. 1. Structure-property correlation. Environmental Science & Technology 35:2536-2542. (i)

Koptsik, G.N. 2014. Problems and prospects concerning the phytoremediation of heavy metal polluted soils: a review. Eurasian Soil Science 47:923-939. (i)

Krutz, L.J., S.A. Senseman, R.M. Zablotowicz, and M.A. Matocha. 2005. Reducing herbicide runoff from agricultural fields with vegetative filter strips: a review. Weed Science 53:353-367. (i)

Kuglerová, L., A. Ågren., R. Jansson, and H. Laudon. 2014. Towards optimizing riparian buffer zones: ecological and biogeochemical implications for forest management. Forest Ecology and Management 334:74-84. (i)

Kümmerer, K. 2009a. Antibiotics in the aquatic environment—a review—Part I. Chemosphere 75:417-434. (i)

Kümmerer, K. 2009b. Antibiotics in the aquatic environment—a review—Part II. Chemosphere 75:435-441. (i)

Lacas, J., M. Voltz, V. Gouy, N. Carluer, and J. Gril. 2005. Using grassed strips to limit pesticide transfer to surface water: a review. Agronomy for Sustainable Development 25:253-266. (i)

Laetz, C.A., D.H. Baldwin, V. Hebert, J.D. Stark, and N.L. Scholz. 2013. Interactive neurobehavioral toxicity of diazinon, malathion, and ethopop to juvenile Coho Salmon. Environmental Science & Technology 47:2925-2931. (i)

Laghlimi, M., B. Baghdad, H. El Hadi, and A. Bouabdli. 2015. Phytoremediation mechanisms of heavy metal contaminated soils: a review. Open Journal of Ecology 5:375-388. (i)

Lamar, R.T., J.W. Evans, and J.A. Glaser. 1993. Solid-phase treatment of a pentachlorophenol-contaminated soil using lignin-degrading fungi. Environmental Science & Technology 27:2566-2571. (i)

Landmeyer, J.E. 2011. Introduction to phytoremediation of contaminated groundwater: historical foundation, hydrologic control, and contaminant remediation. Springer Science& Business Media, New York. (i)

Lee, K.H., T.M. Isenhart, and R.C. Schultz. 2003. Sediment and nutrient removal in an established multi-species riparian buffer. Journal of Soil and Water Conservation 58:1-8. (i)

Lee, P., C. Smith, and S. Boutin. 2004. Quantitative review of riparian buffer width guidelines from Canada and the United States. Journal of Environmental Management 70:165-180. (i)

Leeds, R., L.C. Brown, M.R. Sulc, and L. VanLieshout. 1994. Vegetative filter strips: application, installation and maintenance. Publication number AEX-467-94. Ohio State University Extension, Columbus, Ohio. (viii)

LeFevre, G.H., R.M. Hozalski, and P.J. Novak. 2012. The role of biodegradation in limiting the accumulation of petroleum hydrocarbons in raingarden soils. Water Research 46:6753-6762. (i)

Lewandowski, J., L. Angermann, G., Nutzmann, and J.H. Fleckenstein. 2011. A heat pulse technique for the determination of small-scale flow directions and flow velocities in the streambed of sand-bed streams, Hydrological Processes 25:3244-3255. (i)

Liu, X., X. Zhang, and M. Zhang. 2008. Majors factors influencing the efficacy of vegetated buffers on sediment trapping: a review and analysis. Journal of Environmental Quality 37:1667-1674. (i)

Lortz, H.W., and B.P. McPherson. 1977. Effects of copper and zinc on smoltification of Coho Salmon. EPA-600/3-77-032. U.S. Environmental Protection Agency, Corvallis, Oregon. (i)

Lowrance, R., R. Todd, J. Fail, Jr., O. Hendrickson, Jr., R. Leonard, and L. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34:374-377. (i)

Luo, Y.L., W.S. Guo, H.H. Ngo, L.D. Nghiem, F.I. Hai, J. Zhang, S. Liang, and X.C.C. Wang. 2014. A review on the occurrence of micropollutants in the aquatic environment and their fate and removal during wastewater treatment. Science of the Total Environment 473:619-641. (i)

Lyons, J., S.W. Trimble, and L.K. Paine. 2000. Grass versus trees: managing riparian areas to benefit streams of central North America. Journal of the American Water Resources Association 36:919-930. (i)

Mahar, A., W. Ping, L.I. Ronghua, and Z. Zengqiang. 2015. Immobilization of lead and cadmium in contaminated soil using amendments: a review. Pedosphere 25:555-568. (i)

Malanson, G.P. 1993. Riparian landscapes. Cambridge studies in ecology. Cambridge University Press, Cambridge, United Kingdom. (i)

Mann, R.M., R.V. Hyne, C.B. Choung, and S.P. Wilson. 2009. Amphibians and agricultural chemicals: review of the risks in a complex environment. Environmental Pollution 157:2903–2927. (i)

Marco, A., C. Quilchano, and A. Blaustein. 1999. Sensitivity to nitrate and nitrite in pond-breeding amphibians from the Pacific Northwest, USA. Environmental Toxicology and Chemistry 18:2836-2839. (i)

Mayer, P.M., S.K. Reynolds, M.D. McCutchen, and T.J. Canfield. 2006. Riparian buffer width, vegetative cover, and nitrogen removal effectiveness: a review of current science and regulations. EPA/600/R-05/118. U.S. Environmental Protection Agency, Cincinnati, Ohio. (viii)

Mayer, P.M., S.K. Reynolds, M.D. Marshall, D. McCutchen, and T.J. Canfield. 2007. Meta-analysis of nitrogen removal in riparian buffers. Journal of Environmental Quality 36:1172-1180. (i)

McClain, M.E., E.W. Boyer, C.L. Dent, S.E. Gergel, N.B. Grimm, P.M. Groffman, S.C. Hart, J.W. Harvey, C.A. Johnston, and E. Mayorga. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6:301-312. (i)

McInnis, M.L., and J. McIver. 2001. Influence of off-stream supplements on stream banks of riparian pastures. Journal of Range Management 54:648-652. (i)

McIntyre, J.K., D.H. Baldwin, D.A. Beauchamp, and N.L. Scholz. 2012. Low level copper exposure increases visibility and vulnerability of juvenile Coho Salmon to Cutthroat Trout predators. Ecological Applications 22:1460-1471. (i)

Menichino, G.T., A.S. Ward, and E.T. Hester. 2014. Macropores as preferential flow paths in meander bends. Hydrological Processes 28:482-495. (i)

Merritt, W.S., R.A. Letcher, and A.J. Jakeman. 2003. A review of erosion and sediment transport models. Environmental Modelling & Software 18:761-799. (i)

Misra, A.K., J.L. Baker, S.K. Mickelson, and H. Shang. 1996. Contributing area and concentration effects on herbicide removal by vegetative buffer strips. Transactions of the American Society of Agricultural Engineers 39:2105-2111. (i)

Mitchell, S.M., J.L. Ullman, A.L. Teel, and R.J. Watts. 2014. pH and temperature effects on the hydrolysis of three - lactam antibiotics: ampicillin, cefalotin and cefoxitin. Science of the Total Environment 466-467:547-555. (i)

Mitchell, S.M., J.L. Ullman, A.L. Teel, and R.J. Watts. 2015. Hydrolysis of amphenicol and macrolide antibiotics: chloramphenicol, florfenicol, spiramycin, and tylosin. Chemosphere 134:504-511. (i)

Moermond, C.T.A., F. Roozen, J.J.G. Zwolsman, and A.A. Koelmans. 2004. Uptake of sediment-bound bioavailable polychlorobiphenyls by Benthivorous Carp (*Cyprinus carpio*). Environmental Science & Technology 38:4503-4509. (i)

Monteiro, S.M., J.M. Mancera, A. Fontainhas-Fernandes, and M. Sousa. 2005. Copper induced alterations of biochemical parameters in the gill and plasma of

Oreochromis niloticus. Comparative Biochemistry and Physiology C—Toxicology & Pharmacology 141:375-383. (i)

Moore, M.T., R. Kröger, and C.R. Jackson. 2011. The role of aquatic ecosystems in the elimination of pollutants. Pages 288-304 *in* F. Sánchez-Bayo, P.J. van den Brink, and R.M. Mann, editors. Ecological impacts of toxic chemicals. Bentham Science, Oxford, Mississippi.

Moore, A., N. Lower, I. Mayer, and L. Greenwood. 2007. The impact of a pesticide on migratory activity and olfactory function in Atlantic Salmon (*Salmo salar L.*) smolts. Aquaculture 273:350-359. (i)

Morace, J.L., and S.W. McKenzie. 2002. Fecal-indicator bacteria in the Yakima River Basin, Washington—an examination of 1999 and 2000 synoptic-sampling data and their relation to historic data. Water Resources Investigations Report 02-4054. U.S. Geological Survey, Portland, Oregon. (viii)

Mueller, S.O. 2004. Xenoestrogens: mechanisms of action and detection methods. Analytical and Bioanalytical Chemistry 378:582-597. (i)

Muñoz-Carpena, R., J.E. Parsons, and J.W. Gilliam. 1999. Modeling hydrology ad sediment transport in vegetative filter strips. Journal of Hydrology 214:111-129. (i)

Muscutt, A.D., G.L. Harris, S.W. Bailey, and D.B. Davies. 1993. Buffer zones to improve water-quality—a review of their potential use in UK agriculture. Agriculture Ecosystems & Environment 45:59-77. (i)

Newcombe, C.P., and D.D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11:72-82. (i)

Nimick, D.D., C.H. Gammons, and S.R. Parker. 2011. Diel biogeochemical processes and their effect on the aqueous chemistry of streams: a review. Chemical Geology 283:3-17. (i)

Norris, V. 1993. The use of buffer zones to protect water quality: a review. Water Resources Management 7:257-272. (i)

NRC (National Research Council). 2002. Riparian areas: functions and strategies for management. National Academy Press, Washington, D.C. (i)

Osborne, L.L., and D.A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243-258. (i)

Peterjohn, W.T., and D.L. Correll. 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65:1466-1475. (i)

Peverill, K.I., L.A. Douglas, and N.B. Greenhill. 1977. Leaching losses of applied P and S from undisturbed cores of some Australian surface soils. Geoderma 19:91-96. (i)

Pilon-Smits, E. 2005. Phytoremediation. Annual Review of Plant Biology 56:15-39. (i)

Pinto, E., T.C.S. Sigaud-Kutner, M.A.S. Leitão, O.O. Okamoto, D. Morse, and P. Colepicolo. 2003. Heavy metal-induced oxidative stress in algae. Journal of Phycology 39:1008-1018. (i)

Polyakov, V., A. Fares, and M.H. Ryder. 2005. Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: a review. Environmental Reviews 13:129-144. (i)

PSAT (Puget Sound Action Team). 2007. 2007 Puget Sound update: ninth report of the Puget Sound Ambient Monitoring Program. Puget Sound Action Team, Olympia, Washington. (viii)

Pulford, I.D., and C. Watson. 2003. Phytoremediation of heavy metal-contaminated land by trees—a review. Environment International 29:529-540. (i)

Rahman, F., K.H. Langford, M.D. Scrimshaw, and J.N. Lester. 2001. Polybrominated diphenyl ether (PBDE) flame retardants. Science of the Total Environment 275:1-17. (i)

RCT (R Core Team). 2013. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. (viii)

Redding, J.M., C.B. Schreck, and F.H. Everest. 1987. Physiological effects on Coho Salmon and steelhead of exposure to suspended solids. Transactions of the American Fisheries Society 116:737-744. (i)

Redick, M.S., and T.W. La Point. 2004. Effects of sublethal copper exposure on behavior and growth of *Rana pipiens* tadpoles. Bulletin of Environmental Contamination and Toxicology 72:706-710. (i)

Reichenberger, S., M. Bach, A. Skitschak, and H. Frede. 2007. Mitigation strategies to reduce pesticide inputs into ground- and surface water and their effectiveness; a review. Science of the Total Environment 384:1-35. (i)

Relyea, R.A., and N. Diecks. 2008. An unforeseen chain of events: lethal effects of pesticides on frogs at sublethal concentrations. Ecological Applications 18:1728-1742. (i)

Reuter, J.H., and E.M. Perdue. 1977. Importance of heavy metal-organic matter interactions in natural waters. Geochimica et Cosmochimica Acta 41:325-334. (i)

Rinklebe, J., and S.M. Shaheen. 2015. Miscellaneous additives can enhance plant uptake and affect geochemical fractions of copper in a heavily polluted riparian grassland soil. Ecotoxicology and Environmental Safety 119:58-65. (i)

Rougier, F., D. Troutaud, A. Ndoye, and P. Deschaux. 1994. Nonspecific immune-response of Zebrafish, *Brachydanio rerio* (Hamilton-Buchanan) following copper and zinc exposure. Fish & Shellfish Immunology 4:115-127. (i)

Salt, D.E., R.D. Smith, and I. Raskin. 1998. Phytoremediation. Annual Review of Plant Physiology and Plant Molecular Biology 49:643-668. (i)

Sandahl J.F., D.H. Baldwin, J.J. Jenkins, and N.L. Scholz. 2007. A sensory system at the interface between urban stormwater runoff and salmon survival. Environmental Science & Technology 41:2998-3004. (i)

Sarma, H. 2011. Metal hyperaccumulation in plants: a review focusing on phytoremediation technology. Journal of Environmental Science and Technology 4:118-138. (i)

Schellinger, G.R., and J.C. Clausen. 1992. Vegetative filter treatment of dairy barnyard runoff in cold regions. Journal of Environmental Quality 21:40-45. (i)

Schnaebel, R.R., J.A. Shaffer, W.L. Stout, and L.F. Cornish. 1997. Denitrification distributions in four valley and ridge riparian ecosystems. Environmental Management 21:283-290. (i)

Schultz, R.C., J.P. Colletti, T.M. Isenhart, C.O. Marquez, W.W. Simpkins, and C.J. Ball. 2000. Riparian forest buffer practices. Pages 189-281 *in* H.E. Garett, W.J. Rietveld, and R.F. Fisher, editors. North American agroforestry: an integrated science and practice. American Society of Agronomy, Madison, Wisconsin. (viii)

Schuster, S., and M.E. Grismer. 2004. Evaluation of water quality projects in the Lake Tahoe Basin. Environmental Monitoring and Assessment 90:225-242. (i)

Scoggins, M., N.L. McClintock, L. Gosselink, and P. Bryer. 2007. Occurrence of polycyclic aromatic hydrocarbons below coat-tar-sealed parking lots and effects on stream benthic macroinvertebrate communities. Journal of the North American Benthological Society 26:694-707. (i)

Sen, T.K., and K.C. Khilar. 2006. Review on subsurface colloids and colloid-associated contaminant transport in saturated porous media. Advances in Colloid and Interface Science 119:71-96. (i)

Sheridan, J.M., R. Lowrance, and D.D. Bosch. 1999. Management effects on runoff and sediment transport in riparian forest buffers. Transactions of the American Society of Agricultural Engineers 42:55-64. (i)

Skłodowski, M., E. Kiedrzynska, M. Kiedrzynski, M. Urbaniak, K.M. Zielinska, J.K. Kurowski, and M. Zalewski. 2014. The role of riparian willows in phosphorus accumulation and PCB control for lotic water quality improvement. Ecological Engineering 70:1-10. (i)

Sorensen, E.M. 1991. Metal poisoning in fish. CRC Press, Boca Raton, Florida. (i)

Sprague, J.B., and A. Ramsay. 1965. Lethal levels of mixed copper-zinc solutions for juvenile salmon. Journal of the Fisheries Research Board of Canada 22:425-432. (i)

Spromberg, J.A., D.H. Baldwin, S.E. Damm, J.K. McIntyre, M. Huff, C.A. Sloan, B.F. Anulacion, J.W. Davis, and N.L. Scholz. 2016. Coho Salmon spawner mortality in western US urban watersheds: bioinfiltration prevents lethal storm water impacts. Journal of Applied Ecology 53:398–407. (i)

Straffelini, G., R. Ciudin, A. Ciotti, and S. Gialanella. 2015. Present knowledge and perspectives on the role of copper in brake materials and related environmental issues: a critical assessment. Environmental Pollution 207:211-219. (i)

Susarla, S., V.F. Medina, and S.C. McCutcheon. 2002. Phytoremediation: an ecological solution to organic chemical contamination. Ecological Engineering 18:647-658. (i)

Sweeney B.W., T.L. Bott, J.K. Jackson, L.A. Kaplan, J.D. Newbold, L.J. Standley, W.C. Hession, and R.J. Horwitz. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proceedings of the National Academy of Science 101:14132-14137. (i)

Sweeney, B.W., and J.D. Newbold. 2014. Streamside buffer width needed to protect stream water quality, habitat, and organisms: a literature review. Journal of the American Water Resources Association 50:560-584. (i)

Syversen, N., L. Oygarden, and B. Salbu. 2001. Cesium-134 as a tracer to study particle transport processes within a small catchment with a buffer zone. Journal of Environmental Quality 30:1771-1783. (i)

Tabacchi, E., L. Lambs, H. Guilloy, A. Planty-Tabacchi, E. Muller, and H. Décamps. 2000. Impacts of riparian vegetation on hydrological processes. Hydrological Processes 14:2959-2976. (i)

Tate, K.W., E.R. Atwill, J.W. Bartolome and G. Nader. 2006. Significant *Escherichia coli* attenuation by vegetative buffers on annual grasslands. Journal of Environmental Quality 35:795-805. (i)

Tobiason, S.A., C. Nickerson, R. Zisette, and M. Samadpour. 2002. Stormwater bacteria source tracing at Seattle-Tacoma International Airport. Proceedings of the Water Environment Federation's technical exhibition and conference. Water Environment Federation, Alexandria, Virginia. (viii)

Trimble, G.R., and R.S. Sartz. 1957. How far from a stream should a logging road be located? Journal of Forestry 55:339-341. (i)

Ucar, T., and F.R. Hall. 2001. Windbreaks as a pesticide drift mitigation strategy: a review. Pest Management Science 57:663-675. (i)

UKEA (United Kingdom Environmental Agency). 2009. The hyporheic handbook: a handbook on the groundwater–surface water interface and hyporheic zone for environment managers. Science report SC050070. Integrated Catchment Science Programme, Bristol, United Kingdom. (viii)

Unc, A., and M.J. Goss. 2004. Transport of bacteria from manure and protection of water resources. Applied Soil Ecology 25:1-18. (i)

NRCS (National Resources Conservation Service). 2000. Conservation buffers to reduce pesticide losses. Washington, D.C. (i)

USEPA (U.S. Environmental Protection Agency). 1980. Ambient water quality criteria for copper. EPA 440/5-80-036. Washington D.C. (viii)

USEPA (U.S. Environmental Protection Agency). 2003. Evaluation of sediment transport models and comparative application of two watershed models. EPA/600/R-03/139. Cincinnati, Ohio. (viii)

USEPA (U.S. Environmental Protection Agency). 2015. Washington causes of impairment for 303(d) listed waters. Available: http://iaspub.epa.gov/tmdl_waters10/attains_state.control?p_state=WA#causes/. (January 2019). (viii)

USGS (U.S. Geological Survey). 2006. Pesticides in the nation's streams and ground water 1992-2001 — a summary. Fact Sheet 2006-3028. Sacramento, California . (i)

Vervoort, R.W., D.E. Radcliffe, and L.T. West. 1999. Soil structure development and preferential solute flow. Water Resources Research 35:913-928. (i)

Vidon, P., C. Allan, D. Burns, T.P. Duval, N. Gurwick, S. Inamdar, R. Lowrance, J. Okay, D. Scott, and S. Sebestven. 2010. Hot spots and hot moments in riparian zones: potential for improved water quality management. Journal of the American Water Resources Association 46:278-298. (i)

Vidon, P., and P.E. Cuadra. 2011. Phosphorus dynamics in tile-drain flow during storms in the US Midwest. Agricultural Water Management 98:532-540. (i)

Vogt, T., M. Schirmer, and O.A. Cirpka. 2012. Investigating riparian groundwater flow close to a losing river using diurnal temperature oscillations at high vertical resolution. Hydrology and Earth System Sciences 16:473-487. (i)

Vought, L.B.-M., J. Dahl, C.L. Pedersen, and J.O. Lacoursière. 1994. Nutrient retention in riparian ecotones. Ambio 23:343-348. (i)

Vought. L.B.-M., G. Pinay, A. Fuglsang, and C. Ruffinoni. 1995. Structure and function of buffer strips from a water quality perspective in agricultural landscapes. Landscape and Urban Planning 31:323-331. (i)

Walker, S.E., S. Mostaghimi, T.A. Dillaha, and F.E. Woeste. 1990. Modeling animal waste management practices—impacts on bacteria levels in runoff from agricultural lands. Transactions of the American Society of Agricultural Engineers 33:807-817. (i)

Wang, Z., Z. Wang, and Y. Pei. 2014. Nitrogen removal and microbial communities in a three-stage system simulating a riparian environment. Bioprocess and Biosystem Engineering 37:1105-1114. (i)

Waters, T.F. 1995. Sediment in streams: sources, biological effects and control. American Fisheries Society Monograph 7. American Fisheries Society, Bethesda, Maryland. (i)

Welsch, D.J. 1991. Riparian forest buffers: function and design for protection and enhancement of water resources. NA-PR-07-91. U.S. Forest Service, Radnor, Pennsylvania. (viii)

Welsh, H.H., and L.M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: A case study from California's Redwoods. Ecological Applications 8:1118-1132. (i)

Wilhere, G.F., and T. Quinn. 2018. How wide is wide enough?: values, science, and law in riparian habitat conservation. Natural Resources Journal 58:279-318. (i)

Williams, D.G., and R. Scott. 2009. Vegetation-hydrology interactions: dynamics of riparian plant water use. Pages 37-56 *in* J. Stromberg and B. Tellman, editors. Ecology and conservation of the San Pedro River. University of Arizona Press, Tucson, Arizona. (i)

Wroblicky, G.J., M.E. Campana, H.M. Valett, and C.N. Dahm. 1998. Seasonal variation in surface-subsurface water exchange and lateral hyporheic area of two stream-aquifer systems. Water Resources Research 34:317-328. (i)

Young, R.A., T. Huntrods, and W. Anderson. 1980. Effectiveness of vegetated buffer strips in controlling pollution from feedlot runoff. Journal of Environmental Quality 9:483-487. (i)

Yuan, Y., R.L. Bingner, and M.A. Locke. 2009. A review of effectiveness of vegetative buffers on sediment trapping in agricultural areas. Ecohydrology 2:321-336. (i)

Zhang, Z. 2014. Nonpoint source and water quality monitoring. Pages 261-299 *in* S. Eslamian, editor. Handbook of engineering hydrology: environmental hydrology and water management. CRC Press, Boca Raton, Florida. (i)

Zhang, M., B. Gao, J. Fang, A.E. Creamer, and J.L. Ullman. 2014. Self-assembly of needle-like layered double hydroxide (LDH) nanocrystals on hydrochar: characterization and phosphate removal ability. RSC Advances 4:28171-28175. (i)

Zhang, X., X. Liu, M. Zhang, and R.A. Dahlgren. 2010. A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. Journal of Environmental Quality 39:76-84. (i)

Zhongmin, J., C. Hua, W. Hong, and L. Changxiao. 2012. Photosynthetic responses of the riparian Salix variegate to cadmium stress in Three Gorges Reservoir region. Scientia Silvae Sinicae 48:152-158. (i)

Okanogan National Forest/Scott Fitkin, WDFW

Chapter 6. Nutrient Dynamics in Riparian Ecosystems

By: Paul M. Mayer, Jana E. Compton, and George F. Wilhere

6.1. Introduction

Riparian ecosystems play a vital role in the dynamics of nutrients within watersheds, especially for the three primary macronutrients carbon (C), nitrogen (N), and phosphorus (P). Because they are positioned at the interface between the stream and upland areas, riparian ecosystems mediate the movement of materials and energy between land and water. Riparian areas can be sources or sinks of energy, water, nutrients, organic matter, and organisms within a watershed. Organic matter from riparian areas, an important source of energy and nutrients, makes its way into streams via plant litterfall, or through transport by water, wind, or animals. Organic matter in streams provides habitat and food for microbes, insects, fish, amphibians, birds, and other organisms, and decomposes to release plant-available inorganic nutrients like ammonium, nitrate, and phosphate. Riparian areas also store energy and nutrients from organic matter coming from upland and instream sources through biotic uptake, sorption and exchange, and slowing or trapping particles. Riparian areas also are sites with high rates of nutrient transformation, including conversion of inorganic nitrogen into the gases N_2 or N_20 via denitrification. Riparian ecosystems are influenced by upland activities that supply nutrients that follow flow paths through these ecosystems, eventually reaching streams via surface and subsurface flows. Age, vegetation composition, topography, soil type, and condition of the riparian ecosystem all factor into the fate and transport of nutrients. In addition, the position in the drainage network and connectivity of the riparian area, along with disturbance and management history, affect nutrient processing rates.

Nutrients and the hydrological and biogeochemical processes that dictate their transport and fate are the subject of much current research in riparian ecology (Mayer et al. 2010a, 2014) because of their critical importance for growth and maintenance of life in the riparian ecosystem and the subsequent effects on stream biota and water quality. Although many micro- and macro-nutrients cycle through riparian areas, this chapter addresses only the three primary macronutrients C, N, and P.

Streams in the Pacific Coastal Ecoregion of Washington State are generally oligotrophic, well shaded, and sensitive to nutrient enrichment (Welch et al. 1998). When more N and P enters a riparian ecosystem than can be immediately utilized or stored, degradation of aquatic habitat conditions can occur and negative consequences within or beyond the riparian ecosystem may result. Excess N and P input to streams and estuaries may lead to harmful algal blooms, hypoxia, fish kills, and contamination of drinking water supplies. Although stream nutrient concerns have a somewhat lower profile in the Pacific Northwest (PNW) relative to other areas of the US, such as the Chesapeake Bay or Great Lakes watersheds, approximately 20% of assessed rivers and 8% of assessed lakes in Washington are regarded by the state as having impaired conditions related to an overabundance of nutrients (USEPA ATTAINS database).

Here, we describe nutrient cycles in riparian areas in the PNW, focusing on C, N, and P from natural and anthropogenic sources. Other nutrients, such as calcium, sulfur, or molybdenum, are not as widely studied but have been shown to be important in areas of the PNW for forest growth or other ecosystem processes (Silvester 1989; Chappell et al. 1991; Perakis et al. 2006). When possible, we discuss implications of common land management activities on nutrient dynamics by contrasting disturbed versus undisturbed riparian ecosystems. We also discuss new information gained from recent stable isotope studies and meta-analyses of riparian nutrient removal that have greatly improved our understanding of the permanence and effectiveness of nutrient removal processes in PNW riparian ecosystems.

6.1.1. Nutrient Dynamics in the Pacific Northwest

Figure 6.1 illustrates processes occurring in a largely undisturbed PNW forested riparian ecosystem and associated stream and upland. Water traveling through the riparian area (surface and subsurface flow) is the main transport mechanism for dissolved C, N, and P. These nutrients can be taken up directly by primary producers and decomposers, passively transported to groundwater or the hyporheic zone, or carried to surface waters via subsurface, overland, or channelized flow. The hydrologic regime dictates the timing and magnitude of nutrient movement into aquatic systems, and can create instream pulses of C, N, and P both in particulate and dissolved form (Kaushal et al. 2010).

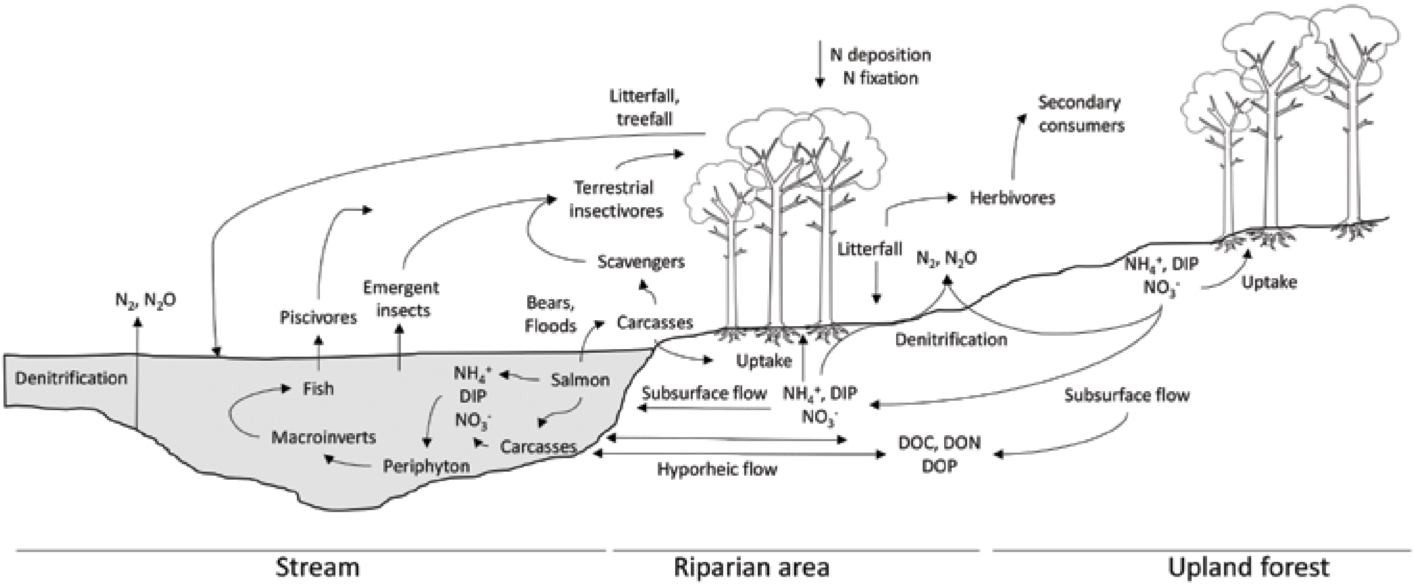

Figure 6.1. Nutrient cycle of an undisturbed PNW forested riparian ecosystem and connected stream and upland forest. Modified from Moyle et al. (2009). DIP = dissolved inorganic phosphorus; DOC, DON, DOP = dissolved organic carbon, nitrogen and phosphorus, respectively.


Most streams in the Pacific Northwest are nutrient-poor ecosystems/Ned Pittman, WDFW

Soil compaction and impervious surfaces typically reduce retention capacity of nutrients in space and time by preventing infiltration and thereby limit the interaction of nutrients with soils, roots, and soil biota. Riparian ecosystems exhibit a continuum of physical characteristics which have implications for nutrient dynamics. Riparian ecosystems along low-order streams, for example, are often narrower and steeper than those along mainstem rivers with wide, level floodplains. Consequently, surface and subsurface hydrological regimes controlling biogeochemical processes in riparian areas also differ along a continuum. Along small, steep forested streams, the water table is usually low and aerobic processes dominate, whereas level areas adjacent to larger rivers may provide water table conditions favoring anaerobic processes that reduce N via denitrification, a microbial transformation of dissolved nitrate N to gaseous molecular N (Burt et al. 2002; Hefting et al. 2004). The hyporheic zone is an area beneath the stream and riparian area where surface water and ground water continually interact. This zone can have a large influence on chemical characteristics of the surface water and, therefore, plays a role in determining biological attributes of streams (Wondzell et al. 2011).

As nutrient molecules move downstream, they may be taken up by algae and microbes living on rocks and wood in the channel in slick mats called biofilms (Cardinale et al. 2002). The polysaccharide layers of biofilms are sites of active nutrient turnover where microscopic algae, fungi, and bacteria consume N and P, die and release nutrients that then move further downstream. Microbial communities may be consumed by grazers such as larval aquatic insects or snails, which in turn may be consumed by larger predators such as fishes or birds. As prey and predators are consumed or die, they release nutrients to the aquatic and terrestrial environments through decay and in this way C, N, and P cycle in a process of uptake and release called nutrient spiraling (Newbold et al. 1982). The distance a molecule moves downstream during a complete cycle is its spiraling length, a metric that explicitly relates nutrient utilization to nutrient supply (Newbold et al. 1982). Long spiraling lengths indicate that: 1) nutrient uptake is inefficient because the system is saturated with nutrients and organisms can no longer fully exploit the incoming nutrient loads, 2) conditions are not optimal for uptake, such as when water flow is too fast, or 3) the stream does not support a community of organisms that effectively processes the nutrients.

In anthropogenically undisturbed Pacific Northwest watersheds, nutrients are ultimately derived from atmospheric and geologic sources, while plants play an important role in the initial uptake of N and P from these sources (McClain et al. 1998). Spatial variation in vegetation, geologic substrates, and human activities drive the variation in stream N and P concentrations and fluxes across the PNW (Wigington et al. 1998; Wise and Johnson 2011). The geology and geomorphology of a watershed and stream channel set the conditions for mineral composition of the groundwater reaching the stream because the shape and slope of the channel and banks affect water movement and velocity. Some bedrock types yield high stream P (Ice and Binkley 2003), leading to high regional P variability in riparian ecosystems and streams. Some types of vegetation can fix atmospheric gaseous N via symbiotic relationships with bacteria in their root systems, and in turn produce more biologically available N than they consume, thereby contributing to N loads in watersheds. A prime example is N_2-fixing Red Alder *Alnus rubra*, an important source of dissolved N in streams in western Oregon and Washington (Compton et al. 2003; Wise and Johnson 2011; Greathouse et al. 2014).

Nitrogen may occur in organic forms as microbial, plant, or animal tissue and in dissolved organic forms, or, if inorganic, primarily as nitrate ($N0_3^-$) and ammonium (NH_3^+), which are forms that can be assimilated by organisms into organic molecules. Forests in the PNW generally have fairly tight N and P cycles with little N or P lost or leached from the system. Rather, there are low rates of N and P inputs and outputs, but high rates of internal cycling (Sollins et al. 1980; Compton and Cole 1998). Triska et al. (1984) showed that N inputs to a stream reach in an old-growth forest watershed in Oregon's Cascade Mountains were dominated by dissolved organic N from groundwater (69% of inputs), with litterfall and precipitation playing smaller roles.

Human activities like timber harvest, farming, or urban development alter nutrient cycles (Figure 6.2), producing higher fluxes, shorter storage durations, and higher concentrations of nutrients in subsurface flow, some of which will enter streams. Disturbances through forestry, grazing, agriculture or urbanization can alter the timing and routing of water through a landscape (Tague and Grant 2004), which can influence nutrient uptake and delivery to streams. Agriculture can contribute to excess nutrients in riparian and aquatic ecosystems through increased runoff of N and P from fertilizer and livestock manure (Wise and Johnson 2011). Point sources and urban stormwater runoff are the dominant sources of N within urban areas. For P, geologic materials dominate natural sources, with point sources such as urban runoff, farm manure, and fertilizer important in areas under anthropogenic land uses. Timber harvest increases stream nitrate and to a lesser extent phosphate concentration (Fredricksen et al. 1975; Scrivener 1975; Martin and Harr 1989). These increases generally persist for a short time, although Brown et al. (1973) saw increases in nitrate persist for six years post-timber harvest and burning.

Historically, a major source of allochthonous[1] nutrients and energy-rich carbon compounds was anadromous salmon (Cederholm et al. 1999; Gende et al. 2002). Anadromous, semelparous fish return from the ocean to spawn and die in their natal streams. The mass of every salmon carcass, of which 95% is accumulated in the marine environment (Naiman et al. 2002), is deposited in oligotrophic freshwater environments. The magnitude of nutrient and chemical energy losses to freshwater ecosystems caused by declines in salmon populations can best be appreciated by comparing current and historical salmon run sizes. Gresh et al. (2000)

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[1] Allochthonous means originating or formed in a place other than where found. Hence, in the context of aquatic ecosystems, allochthonous nutrients found in a stream originated in terrestrial ecosystems. In contrast, autochthonous means originating or formed in the place where found.

Historically, carcasses of adult salmon delivered approximately 100,000 tons of nutrients to rivers and streams of Washington/Ned Pittman, WDFW

estimated that the current biomass of salmon runs for all Pacific salmon species (Chinook *Oncorhynchus tshawytscha,* Coho *Oncorhynchus kisutch,* Sockeye *Oncorhynchus nerka,* Chum *Oncorhynchus keta,* and Pink *Oncorhynchus gorbuscha*) in Puget Sound, the Washington Coast, and the Columbia River Basin are 25, 5, and 1% of historical biomass (circa early 1900s), respectively. This represents a loss of 122,470 US tons per year (111,104 metric tons) of marine-derived nutrients and chemical-energy in these watersheds. Naiman et al. (2002) estimated that in the Willapa Bay Watershed the biomass of Chinook, Coho, and Chum Salmon carcasses in streams decreased from 2,920 to 226 US tons (2,649 to 205 metric tons per year), and that marine derived N and P decreased by more than 90%.

Figure 6.2 represents nutrient cycling in a watershed with more human-associated disturbance, including an absence of salmon returns; by comparing Figures 6.1 and 6.2, changes in the cycling loop and intra-system transfers are evident when salmon are missing. Much of the cycling of marine-derived nutrients is mediated by predators and scavengers that move large amounts of marine-derived nutrients substantial distances into the terrestrial environment either as salmon carcasses or as metabolic waste products, i.e., urine and feces (Naiman et al. 2002). In several small streams on the Olympic Peninsula, Cederholm et al. (1989) observed that Coho Salmon carcasses were consumed by 22 species of mammals and fish (e.g., American Black Bear *Ursus americanus,* Northern River Otter *Lontra canadensis,* Mink *Mustela vison,* Bald Eagles *Haliaeetus leucocephalus,* and Ravens *Corvus corax*), and that 40% of salmon carcasses were moved by scavengers from the water to riparian areas. Flooding also can deposit salmon carcasses in riparian areas (Ben-David et al. 1998). Salmon carcasses provide N and P nutrients for microbes, plants, and animals, and influence all trophic levels in riparian ecosystems (Willson and Halupka 1995; Compton et al. 2006). Trees and shrubs in riparian areas along spawning sites in the Kadashan and Indian rivers of southeast Alaska derived approximately 23% of their foliar nitrogen from decayed salmon carcasses, and growth rates of Sitka Spruce *Picea sitchensis* were significantly increased (Helfield and Naiman 2001). On average, 17% of the nitrogen incorporated into the riparian vegetation of a stream in western Washington was derived from carcasses of spawning salmon, and significant percentages of marine-derived N (up to 30%) and C (up to 39%) were found in aquatic insects, Cutthroat Trout *Oncorhynchus clarkii*, and juvenile salmon (Bilby et al. 1996).

Other animals subsidize riverine food webs by transporting nutrients across watersheds or by altering the hydrology (Chapter 8 in this document; Naiman and Rogers 1997; Masese et al. 2015). As ecosystem engineers, American Beaver *Castor canadensis* have a significant influence on stream nutrients. Beaver can influence biogeochemical cycles by impounding streams and changing hydrologic regimes and organic matter and sediment transport (Pollock et al. 2007).

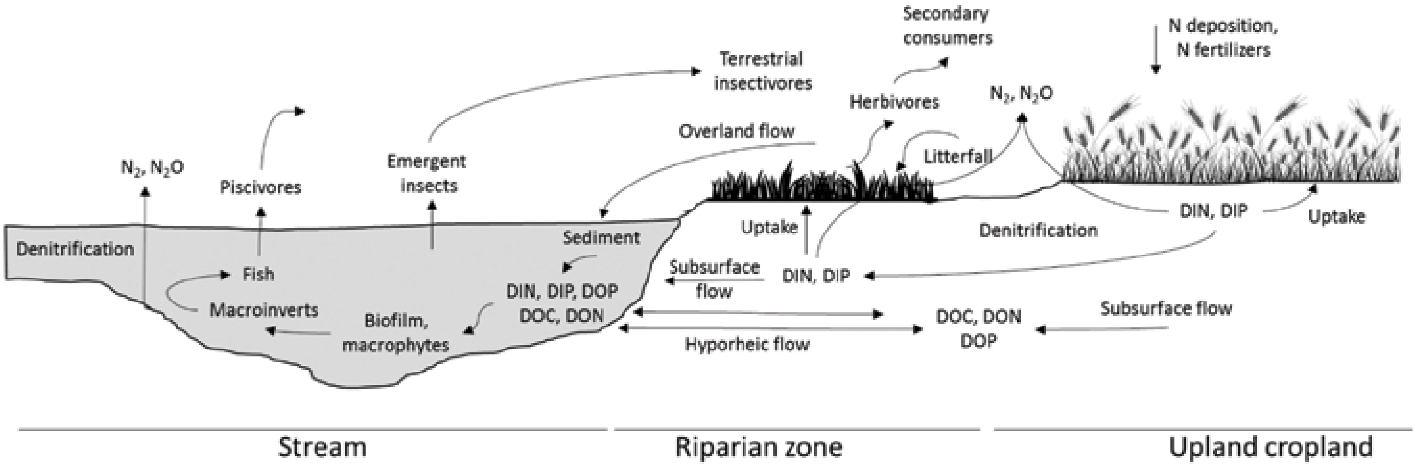


Figure 6.2. Nutrient cycle of a disturbed PNW riparian ecosystem and connected stream and upland cropland. Much of the cycling of marine-derived nutrients is facilitated by predators and scavengers that move nutrients into the terrestrial environment either as salmon carcasses or as urine and feces. Modified from Moyle et al. (2009). DIP = dissolved inorganic phosphorus; DIN = dissolved inorganic nitrogen (nitrate and ammonium); DOC, DON, DOP = dissolved organic carbon, nitrogen and phosphorus, respectively.

Retention of total N and P increased 72% and 43%, respectively after Beaver populations recovered in a Minnesota watershed (Naiman et al. 1994) and organic matter (carbon) retention increased three-fold after a stream was impounded by Beaver (Naiman et al. 1988). However, Beaver impoundments also influence N dynamics by flooding riparian forests, creating anoxic conditions in forest soils leading to an increase in denitrification and subsequent reduction in N (Naiman et al. 1988). Elk *Cervus canadensis*, deer (*Odocoileus* spp.), and livestock can also be important sources of N and P when found in close proximity to streams.

Riparian vegetation can affect uptake and delivery of nutrients to streams. Riparian plant roots can slow water movement and stabilize soils as well as take up dissolved nutrients. The types of vegetation and growth patterns in riparian areas determine the amount and timing of litterfall and woody material delivery to a stream. The height and density of vegetation dictate shading and solar input to the stream, thus affecting conditions for plant growth and N and P uptake. The key limiting nutrients for primary production in PNW streams are N and P (Welch et al. 1998), but there has not been enough experimental work conducted across the region to determine precisely when and where this occurs.

Shallow groundwater adjacent to stream channels often is a hot spot for N removal processes and a storage zone for other solutes.

6.1.2. Carbon

The cycling of C in riparian and aquatic ecosystems is complex and can differ among locations and through time because C can be available in different forms and is produced by different sources, thereby affecting its transport and fate. Carbon is tightly linked to N and P, driving biological transformations of these nutrients. Organic C regulates ecosystem functions in streams and rivers because dissolved organic carbon (DOC) is a primary energy source for microorganisms. Riparian vegetation in forests influences DOC composition and contributes a substantial proportion of stream organic C budgets (McDowell and Likens 1988). DOC in streams is a mixture of recalcitrant, (i.e., difficult to process components) and more easily metabolized labile fractions that drive biogeochemical processes involving nitrogen (Kaushal and Lewis 2005). For example, availability of dissolved and particulate organic C can limit denitrification, a microbial process critical to maintaining water quality, which has important implications for managing N in streams (Newcomer et al. 2012). Denitrification occurs only under anaerobic conditions where microbes shift from respiring oxygen to respiring nitrate, releasing nitrous oxide, and inert N_2 gas. These conditions may be ephemeral, fluctuating in response to water tables and associated changes in redox potential (Mayer et al. 2010b).

Anoxic conditions also promote the production of methane. Production of methane and nitrous oxides, increase in riparian areas with increased organic C supply (Kaushal et al. 2014). Dissolved organic carbon also can increase biological oxygen demand, thereby reducing oxygen for fish (Stanley et al. 2012).

Various organic C types from biofilm, leaves, and soil organic matter may have differential effects on ecosystem functions in forested streams (McDowell and Likens 1988), agricultural streams (Royer and David 2005), and urban streams. For example, C derived from grasses supports more denitrification than C from tree leaves (Newcomer et al. 2012). Because the microbes that influence the transport and fate of N in riparian ecosystems require C as an energy source, C and N are inextricably linked in a biogeochemical sense.

Shading of small headwater streams in forested watersheds can obscure 95% of sunlight (Murphy 1998); hence, in-stream productivity is typically light-limited. Consequently, the main energy source for such streams is allochthonous chemical energy contained in litter from riparian areas (Conners and Naiman 1984; Bisson and Bilby 1998). In headwater streams, litter directly supports an animal community compromised predominantly of invertebrate detrivores which, in turn, support a community of invertebrate predators (Wallace et al. 1997). Detrivores, microbes, and physical breakup convert the coarse particulate organic material (CPOM) of litter to fine particulate organic material (FPOM) which is utilized by invertebrate communities downstream (Bisson and Bilby 1998; Richardson and Danehy 2007). Headwaters comprise 60 to 80% of the cumulative length of a watershed's drainage network (Benda et al. 2005); hence, headwaters contribute a substantial amount of chemical energy to mainstem rivers and associated biota. For example, Wipfli (2005) estimated that every kilometer of fish-bearing stream received enough energy from headwaters to support 100 to 2,000 young-of the-year salmonids.

Carbon cycles through riparian ecosystems via CO_2 uptake from the atmosphere through photosynthesis, litterfall, decomposition and/or consumption. Riparian litter is a critically important source of C for streams as energy for invertebrates and microbes that form the basis of food webs. Deciduous-dominated riparian forests grow quickly and initially deliver more litter to streams and have a more pronounced seasonal litter contribution than coniferous-dominated forests (Hart et al. 2013). Conifers, however, play an important structural role as large wood in streams. Deciduous species like Red Alder can increase fine litter flux and nutrient delivery to terrestrial and aquatic food webs, thus complementing the year-round shade and wood provided by conifers (Hart et al. 2013).

The main energy source for small headwater streams is allochthonous chemical energy provided by leaf litter/Marc Hayes, WDFW

Chemical energy, required by heterotrophic organisms (those that do not produce their own food from sunlight), is contained within the various C compounds of organic matter. The fundamental effects of organic matter on ecosystem composition, structure, and functions have motivated conceptual theories to describe riverine ecosystems (Pingram et al. 2012). The River Continuum Concept (RCC; Vannote et al. 1980) describes the balance of allochthonous C (produced externally to the stream) to autochthonous C (produced in-stream) in riverine ecosystems from headwaters to mainstem rivers. This balance plays a key role in structuring the invertebrate community. According to RCC, in narrow headwater streams (first to third order), nearly all C is allochthonous produced by litter from riparian vegetation because shade severely limits the production of autochthonous C produced through in-stream photosynthesis. Because medium-sized streams (fourth to sixth order) are wider, shading decreases, in-stream photosynthesis increases, and the density of litter inputs decreases. Hence, the quantities of allochthonous and autochthonous C may be about equal. In large rivers (≥ seventh order), shading effects only a small portion of surface water, however, turbidity and water depth limit photosynthetic production. Consequently, according to RCC, the main source of energy in large rivers is allochthonous C from upstream. The RCC also posits that the quantity and quality of organic matter, which changes along the river continuum, affects the composition of the aquatic invertebrate community.

Litter from riparian areas is clearly an important source of limiting nutrients and organic matter for aquatic ecosystems. Consequently, providing adequate amounts of litter to streams is an important issue for riparian area management and addressing this issue requires an understanding of litter delivery into streams. FEMAT (1993:V-26) developed a conceptual model, known as the FEMAT curves, that depicts relationships between riparian ecological functions, including litterfall, and distance from a stream channel. Bilby and Heffner (2016) studied factors influencing litter delivery into streams in young and mature conifer forests of the western Washington Cascades and found that litter travel increased with increasing tree height, topographic slope, and wind speed. They estimated that riparian buffer widths needed to capture 95% of annual litter input from mature conifer forests to streams are between 60 to 83 ft (18 to 25 m) depending on a site's slope and wind exposure. The mean tree height of their three mature forest sites was 154 ft (47 m). Hence, riparian buffer widths needed to meet the 95% capture objective range from 39 to 54% of mean tree height. Bilby and Heffner (2016) warn that they may have underestimated litter travel distances because they released litter from only the bottom of the tree canopy, but litter travels farther with increasing height, and therefore, litter produced within the tree canopy is likely to travel farther.

While stream flows carry organic matter, insects, and other invertebrates downstream to subsidize food webs in higher order streams, there may also be upstream movement of aquatic organisms that subsidizes food webs in lower order streams. In a Welsh mountain stream, for example, amphipods *Gammarus pulex* showed net upstream movement whereas stonefly (*Trichoptera* spp.) females showed no upstream flight preference, but females moving upstream contained twice as many eggs (Dudley-Williams and Williams 1993). In many cases, however, such as with mayflies (family Baetida), seasonal upstream movements of adults via an aerial pathway are pronounced (Hershey et al. 1993). Such upstream movement of nutrients and chemical energy may be important, but has been little studied.

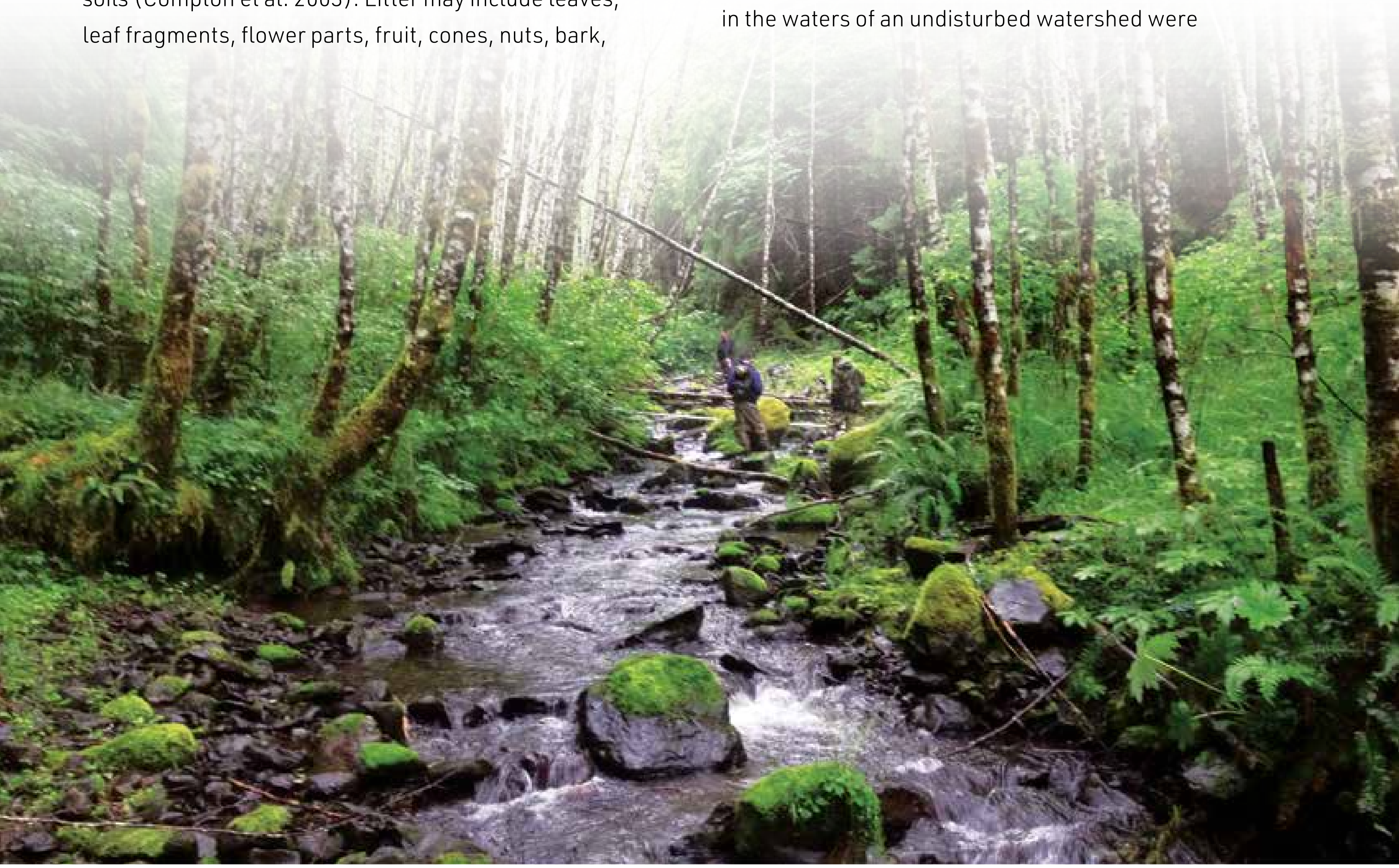
Large wood in stream channels can affect nutrient cycling by slowing the downstream transport of water, sediment, organic matter, and other materials (Chapter 3). Large wood in riparian areas ultimately decomposes, providing nutrients to the forest floor thereby increasing moisture holding capacity (Harmon et al. 1986). Woody material in streams can be a direct source of labile C to streams that can fuel microbial processes, though most C in wood is relatively recalcitrant, taking years or decades to decompose. Wood slows the flow of water thereby extending the residence time of water in the channel and facilitating accumulation of finer organic and inorganic sediments (Chapter 2) that support biofilms.

6.1.3. Nitrogen

Nitrogen is a component of proteins, necessary for all life, and often is a limiting nutrient for both terrestrial and aquatic organisms. N limitation of plant growth is common in PNW forests (Peterson and Hazard 1990). Ultimately, N comes from atmospheric N_2, which has a very strong triple bond that, aside from lightning and industrial processes, only a few specialized N-fixing organisms can break.

Nearly all N in lotic ecosystems of Washington is derived from allochthonous sources, and in undisturbed headwater watersheds, the source of nearly all N is terrestrial. Nitrogen can enter a river or stream via several pathways: litterfall, movement of soluble N from riparian soils into groundwater and hyporheic zones, and leaching and subsurface transport from upland soils (Compton et al. 2003). Litter may include leaves, leaf fragments, flower parts, fruit, cones, nuts, bark, branches and twigs (Benfield 1997), however, leaves account for 72% of total litter in broad-leafed forests and 80% in needle-leaved forests (Xiong and Nilsson 1997). Litter may enter surface waters through direct fall, lateral movement along the ground, or mobilization during floods.

For a headwater stream in the western Cascades of Oregon, Triska et al. (1984) found that more than 90% of annual N inputs to the stream were derived from biotic processes in the adjacent forest. Seventy-three percent of N entering the stream was dissolved organic N in water flowing through the subsurface from forest to stream and 21% was contained in organic matter (e.g., leaf litter, needles, and coarse woody debris) from the riparian forest. The headwater stream exported 74% of its annual N inputs to downstream waters; 22% of the exported N was contained in particulate organic matter. Edmonds et al. (1995) found that nitrate concentrations in the waters of an undisturbed watershed were



N-fixation by Red Alder is an important component of nutrient cycling in forests of western Washington and Oregon/Ned Pittman, WDFW seasonal, with in-stream concentrations being highest in the fall and lowest in the summer when N uptake by terrestrial vegetation was greatest.

Red Alder fixes N_2 and enriches soil with N compounds through root and nodule secretions. Red Alder leaves contain N concentrations roughly 2 to 3.5 times greater than leaves or needles of other tree species (Tarrant et al. 1951). Because aquatic ecosystems in undisturbed watersheds are often N-limited, Red Alder can be a major source of allochthonous N to aquatic ecosystems, and more Red Alder litter can increase ecosystem productivity. For example, Wipfli and Musslewhite (2004) found that headwater streams with more Red Alder canopy cover exported greater biomass of both terrestrial and aquatic invertebrates to fish-bearing waters than riparian areas with less alder. This subsidy of prey from headwaters to fish-bearing streams is needed to maintain fish productivity (Wipfli and Baxter 2010).

Nitrogen cycles in the environment through multiple redox states, in solid, dissolved, and gaseous forms, and originates from multiple sources, making for one of the most challenging biogeochemical cycles to monitor and characterize (Figure 6.3). Shallow groundwater adjacent to stream channels often is a hot spot for N removal processes and a storage zone for other solutes (Hinkle et al. 2001; Zarnetske et al. 2011). The groundwater–surface water interface is characterized by dynamic gradients of dissolved oxygen, N, and organic C concentrations where biogeochemical reactions take place, including metabolism of organic C, denitrification, and nitrification (Sobczak et al. 2003). Denitrification is generally considered the most important biological process through which ecosystems lose N. Denitrification, and disturbances like tree harvest and fire, remove N from an ecosystem, whereas N uptake by plants eventually returns N to the system through litterfall or senescence and subsequent microbial decay. Organic C is required as an energy source for denitrification. Small streams can lose nitrate

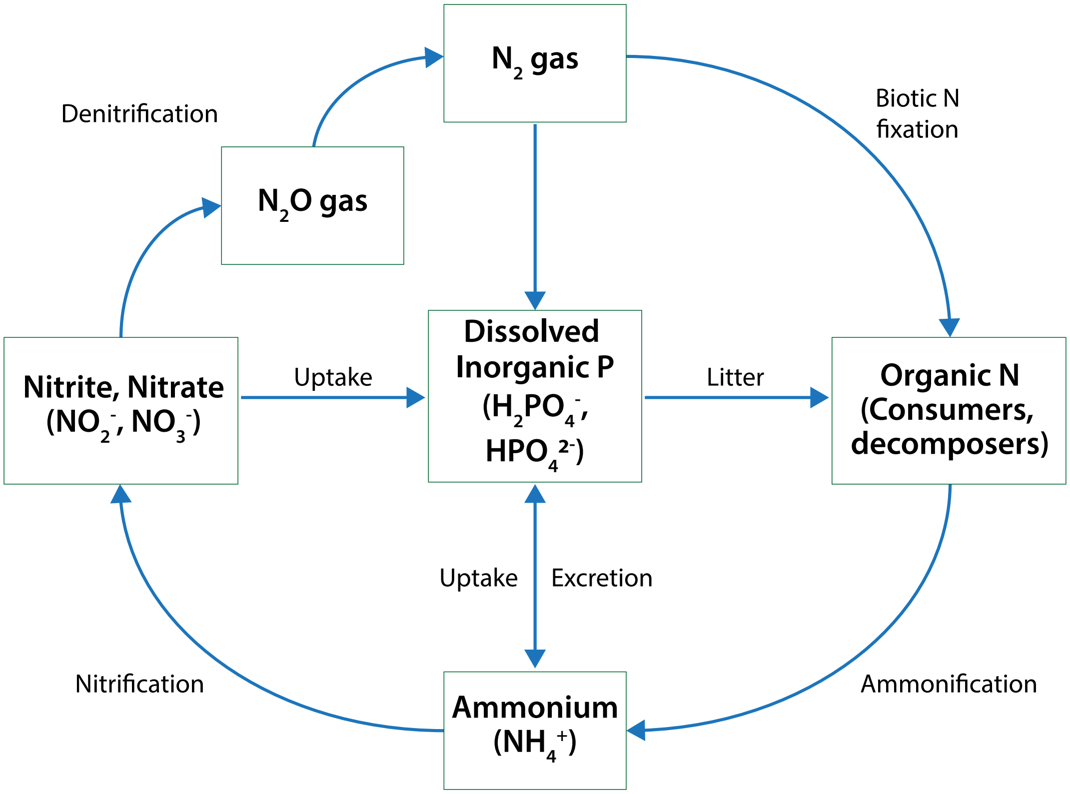


Figure 6.3. Nitrogen cycle pools and processes in ecosystems.

N efficiently because of their high ratio of streambed area to water volume, which brings more surface water in contact with the hyporheic zone. Small streams have a large cumulative influence on N transformation and loss because they account for most of the stream length within a network (Alexander et al. 2000; Peterson et al. 2001). Larger streams lose nitrate N because of longer transport distances over which more biological reactions can occur that consume N coupled with longer water residence times and higher N concentration which fuels the denitrification along with organic C (Mulholland et al. 2008).

Increased groundwater–surface water interaction can alter dissolved oxygen concentrations and redox conditions (Striz and Mayer 2008; Zarnetske et al. 2012), transporting N- and C-containing organic matter to microbes in subsurface sediments, leading to N loss via denitrification under favorable oxidation–reduction conditions (Hedin et al. 1998). The conditions that favor denitrification may change rapidly over time and space. Therefore, "hot spots" may form where denitrification occurs in small patches of the stream (e.g. under leaf packs) or "hot moments" where denitrification rates are high but for short periods of time due to rapid dynamic water movement (McClain et al. 2003; Vidon et al. 2010). However, biochemical processes in urban riparian soils such as denitrification may be impaired because of increased rates of erosion and limited groundwater–surface water interaction due to high proportions of impervious surfaces (Groffman et al. 2002).

The importance of riparian vegetation on transforming N inputs has been examined (Sweeney et al. 2004; Mayer et al. 2007; Sweeney and Newbold 2014). Riparian area characteristics such as width of vegetation, soil carbon, and hydrologic flowpaths, can influence the amount of N removed. Mayer et al. (2007) surveyed the available scientific literature containing data on riparian management and N concentration in streams and groundwater to identify relations between N uptake and riparian width, hydrological flow path, and vegetative cover. Removal efficiencies ranged from <0 (i.e., where the riparian ecosystem appeared to be a N source) to 100%, with a median removal rate of 91%. Wide vegetated riparian areas (>165 ft; 50 m) more consistently removed significantly higher proportions of N entering riparian areas than narrow bands of vegetation (0 - 82 ft; 0 - 25 m). The critical conclusions from this review were that riparian areas were effective at trapping and removing N when they were vegetated, wider, and allowed for subsurface movement of water where the interaction of nutrients dissolved in the water had prolonged contact with soils. Therefore, the hydrologic connection of riparian areas and streams is key to effective nutrient capture. Where channels are incised, riparian areas tiled for drainage, or where floodplains are disconnected via reinforced banks or channelization, nutrients will flow downstream with little interaction with riparian areas.

Subsurface hydrology (saturated vs. unsaturated soil conditions) and redox condition appear to be significant determinants of N-removal efficiency, regardless of vegetation (Mayer et al. 2007; Mayer et al. 2010b). When the water table fluctuates in response to precipitation events, buried relict hydric soil may become saturated, producing sub-oxic or anoxic conditions (Hefting et al. 2004; Weitzman et al. 2014), facilitating N removal. Where flow paths move through wetland riparian soils, great potential exists to remove large quantities of N in proportion to the loads (Jordan et al. 2011); where flows occur via deeper groundwater dynamics or via tile

Riparian vegetation: Red Elderberry Sambucus racemosa/
Ned Pittman, WDFW

drainage, N in solution may bypass the C-rich riparian soils and flow through zones where less potential exists for nutrient removal (Wigington et al. 2005). Hydrologic connectivity between riparian areas and streams is a critical factor in determining N removal rates.

Denitrification rates in the stream channel are linked to the organic C content of benthic sediments, respiration rates, and extent of stream water interaction with the streambed (Alexander et al. 2000; Mulholland et al. 2008), conditions that can be fostered by woody material (e.g., sticks, branches or tree trunks), and debris dams in streams. Woody material may function as microsites or hotspots of elevated biogeochemical cycling including denitrification (Groffman et al. 2005), and provides substrates for bacteria and fungi, which in turn consume N in streams (Ashkenas et al. 2004).

Biofilms are a matrix of algae, bacteria, and fungi embedded in a slick polysaccharide film that coats hard substrates in streambeds. These microbial communities consume, respire, and reduce inorganic N (Mulholland et al. 1995). Physical habitat heterogeneity and the biodiversity of the biofilms influence N uptake (Cardinale et al. 2002; Cardinale 2011). Biofilm structure, composition, and capacity for biogeochemical cycling are influenced by substrate composition, light penetration, nutrient concentration, flow rates, seasonality, sediment composition, and the community of invertebrate grazers in the vicinity (Sabater et al. 2002). Biofilms formed on wood substrates have been found to have higher respiration rates and greater N demand than biofilms developed on rock substrates (Sabater et al. 1998). Lazar et al. (2014) found that biofilms on wood substrates in a forested stream had significantly higher denitrification rates than those on non-organic substrates such as stone. In an old-growth forest system in Oregon, aquatic mosses and biofilm on large wood showed the highest N uptake rates among biota while, both vertebrate and invertebrate consumers took up substantial amounts of N, especially small invertebrate grazers (Ashkenas et al. 2004). Furthermore, N in the stream was transported to the upland through uptake by terrestrial plants, likely through long, subsurface flow paths, demonstrating a strong link between terrestrial and aquatic systems (Ashkenas et al. 2004).

In addition to the biofilms on the streambed, microbial communities within the hyporheic zone can be hotspots for N dynamics. At the HJ Andrews Experimental Forest in Oregon, Zarnetske et al. (2011) found that the hyporheic zone was an important site for N dynamics in small forested streams, with residence time determining the dominance of different N transformation processes. Water pathways with residence times of less than 7 hours tended to be dominated by DO and DOC use as well as ammonium and nitrate production, while pathways with longer residence times removed N via denitrification. Further modeling suggested that water residence times and oxygen uptake rates of microbes are important determinants of whether hyporheic zones are net sources or sinks of nitrate N (Zarnetske et al. 2012). In large rivers like the Willamette, hyporheic zones can be important areas for nitrate removal from regional groundwater prior to entry into surface flow (Hinkle et al. 2001), as well as important for N removal and cooling as water moves in and out of hyporheic areas on its path downstream (Fernald et al. 2006).

The relationships between land use and N transport and fate can be complex, and are mediated by aquatic and riparian biota. Sobota et al. (2012) showed that N uptake differs substantially across forested, agricultural, and urban streams in the Willamette River basin in Oregon. Most nitrate N in forested streams was transported downstream without being taken up. N storage was high in algal biomass in unshaded agricultural and urban streams but turnover and subsequent recycling back to the stream may ultimately be high (Sobota et al. 2012). Collectively, this work indicates that uptake via heterotrophic organisms living on large wood is critical to long-term N capture in forested streams. Once the canopy is opened up, autotrophic algal production becomes a more important process for uptake of excess nutrients, which leads to shorter-term storage and potential for negative consequences associated with freshwater algal blooms.

6.1.4. Phosphorus

Phosphorus (P) is an important component of adenosine triphosphate (ATP) and necessary for energy transformations in plant and animal cells. Like N, low P availability often limits primary production on land and can be particularly limiting to productivity in freshwater. This is because P tends to form complexes and be retained in soils and thus does not move as readily into ground and surface waters from the landscape as does nitrate N. P cycles in the environment through multiple organic and inorganic forms (Figure 6.4). P occurs in the environment in organic forms (e.g., organo-phosphate and polyphosphates like ATP) in plant and animal biomass, sewage, and pesticides. Inorganic forms of P, such as phosphate ($P0_4^{3-}$), and orthophosphate, are biologically reactive and thus can be used directly by plants and microorganisms (Carpenter et al. 1998). Orthophosphate is derived from phosphate-bearing minerals, fertilizers, detergents, and industrial chemicals. In the Pacific Northwest, bedrock is the major source of stream P loads, with fertilizer, manure, and wastewater inputs important in some locations (Wise and Johnson 2011). Overabundance of P can result in abnormally high primary production in water, leading to water quality problems such as algal blooms, production of cyanotoxins, and low oxygen events (Carpenter et al. 1998; Jacoby and Kann 2007).

P is commonly a limiting nutrient in aquatic ecosystems because it is highly insoluable and has an affinity for binding to soil particles. Nevertheless, riparian areas are sources of allochthonous P to streams in undisturbed watersheds. The amount of P that is contributed by riparian areas to aquatic habitats through litterfall

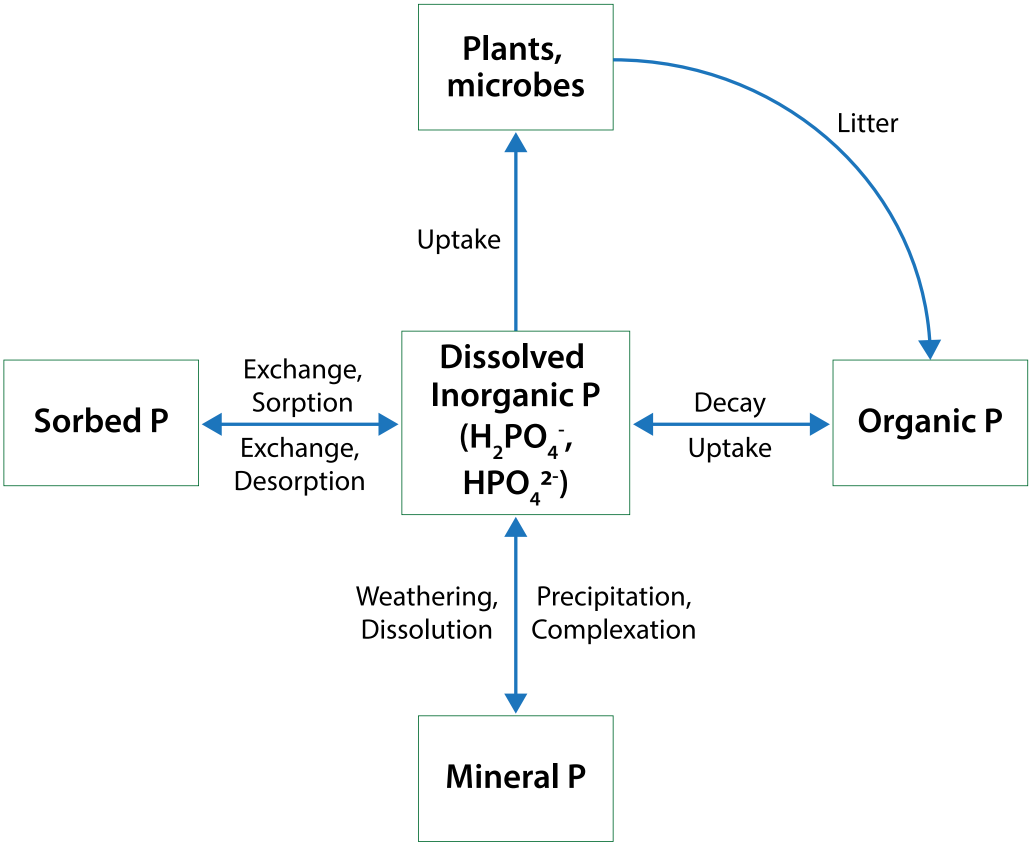


Figure 6.4. Phosphorus cycle pools and processes in ecosystems.

relative to other sources under natural, undisturbed conditions has yet to be quantified for riparian ecosystems in the Pacific Northwest.

P attenuation refers to declines in P concentration in water and soil through physical, chemical, and biological processes. Mineral content and pH are important factors determining P attenuation in soil. Because of their volcanic origins and the weathering environments, many soils in the PNW can bind tightly to large quantities of anions like phosphate, replacing reactive soluble P forms for particulate P forms, which are less biologically reactive (Johnson and Cole 1980; Bohn et al. 1985). Sorption, the chemical attachment of molecules to soil particles, may occur quickly (e.g., days) and includes adsorption and substitution between P and other anions on mineral surfaces. Sorption may also occur slowly (e.g., weeks), such as mineral dissolution and precipitation reactions between P and soil cations such as Ca and Mg or Al, Fe, and Mn oxide compounds depending on soil pH (Bohn et al. 1985). Phosphate is most soluble in slightly acid to neutral pH soils. Under reducing conditions, Fe is reduced and the P-Fe oxide compounds may dissolve, thereby releasing P (Denver et al. 2010). Minimally disturbed forests have a relatively tight P cycle, with low inputs from precipitation, and even lower leaching rates (Yanai 1992; Compton and Cole 1998). In the volcanically influenced soils of the Cascades, most of the ecosystem P is contained in soil, and organic and inorganic forms of sorbed P are the dominant forms as opposed to primary minerals or microbial P (Compton and Cole 1998). There is substantial recycling of P by plants and soil and biotic release of organic P via phosphatase enzyme activities is an important component of this cycle (Giardina et al. 1995). Several studies have shown that Red Alder increases rates of P cycling (Giardina et al. 1995; Zou et al. 1995), further supporting the important role that Red Alder plays in regulating both N and P nutrient cycles of the PNW. Vegetated riparian areas generally are expected to retain P by slowing water flow and by plant and microbial uptake.

Unlike N, which is ultimately constrained by atmospheric fixation and regulated by biologic cycling, P availability is controlled by weathering and a complex set of chemical and biological processes. P retention is regulated by the equilibrium P concentration (EPC), the concentration at which P sorption equals desorption (Hoffmann et al. 2009). Once soil P has reached its sorption capacity, excess P may be exported in water (Domagalski and Johnson 2012). Biological processes also play a role in P attenuation and release (Schechter et al. 2013). Bacteria, fungi, algae, and plants incorporate P into biomass; however, plants vary in P demand and uptake effectiveness, exhibiting numerous strategies for sustaining growth and maintenance (Shen et al. 2011). P is released from organic matter through decomposition, which is dependent on pH, litter quality (C:N:P ratios), Ca content, redox potential, soil moisture, and temperature (Schechter et al. 2013). Increases in P loading as well as soil disturbance and erosion have the potential to reduce this generally tight P cycle and increase transfer to aquatic ecosystems (Figure 6.2).

Microbial and plant uptake represent short-term, transient P pools (Richardson and Marshall 1986). Phosphate may be assimilated into plant tissues or in microorganisms but released upon death and senescence. Peat accumulation and P sorption and precipitation are important long-term P sequestration mechanisms (Richardson and Marshall 1986; Reddy et al. 1995). Long-term storage of P depends on sorption to inorganic sorbents; P sequestration is mainly associated with adsorption to Fe and Al oxides in acidic soils or precipitation of Ca phosphates in alkaline soils (Giesler et al. 2005). Phosphorus retention mechanisms differ widely depending on hydrological pathway through the riparian area. Sorption and desorption reactions are more important during subsurface flow, while sedimentation of particulate P may be the major retention mechanism during overland flow. Significant amounts of P can be stored in stream sediments and may be resuspended and released during storms. Retention of total P in riparian areas is controlled mainly by sedimentation processes, dependent on morphology (e.g., width) and vegetation characteristics. However, riparian areas can become a net source of dissolved P released from soil or from plant material (Hoffman et al. 2009). Retention of dissolved P in riparian areas is often not as significant as retention of particulate P (Hoffman et al. 2009). Inundation of riparian areas or floodplains can deposit particulate P whereas plant uptake may temporarily immobilize P (Hoffman et al. 2009). Overall, riparian ecosystem retention efficiency varies widely and some riparian areas may become sources of P, especially ortho-phosphate (Schechter et al. 2013). Total P moving through the soil is not necessarily related to the width of the riparian area but rather on flow path and soil types (Hoffman et al. 2009; Schechter et al. 2013).

6.2. Hydrologic Connectivity and Nutrient Dynamics

Nutrients cycle at various spatial and temporal scales. Soils, vegetation, and water flows largely dictate the sources and processing of nutrients. Nutrients may enter the riparian ecosystem from the atmosphere (e.g., nitrogen oxides produced from fossil fuel combustion or ammonia released from agricultural areas), from terrestrial sources when rainwater flushes nutrients from upland soils and vegetation, or from floods that

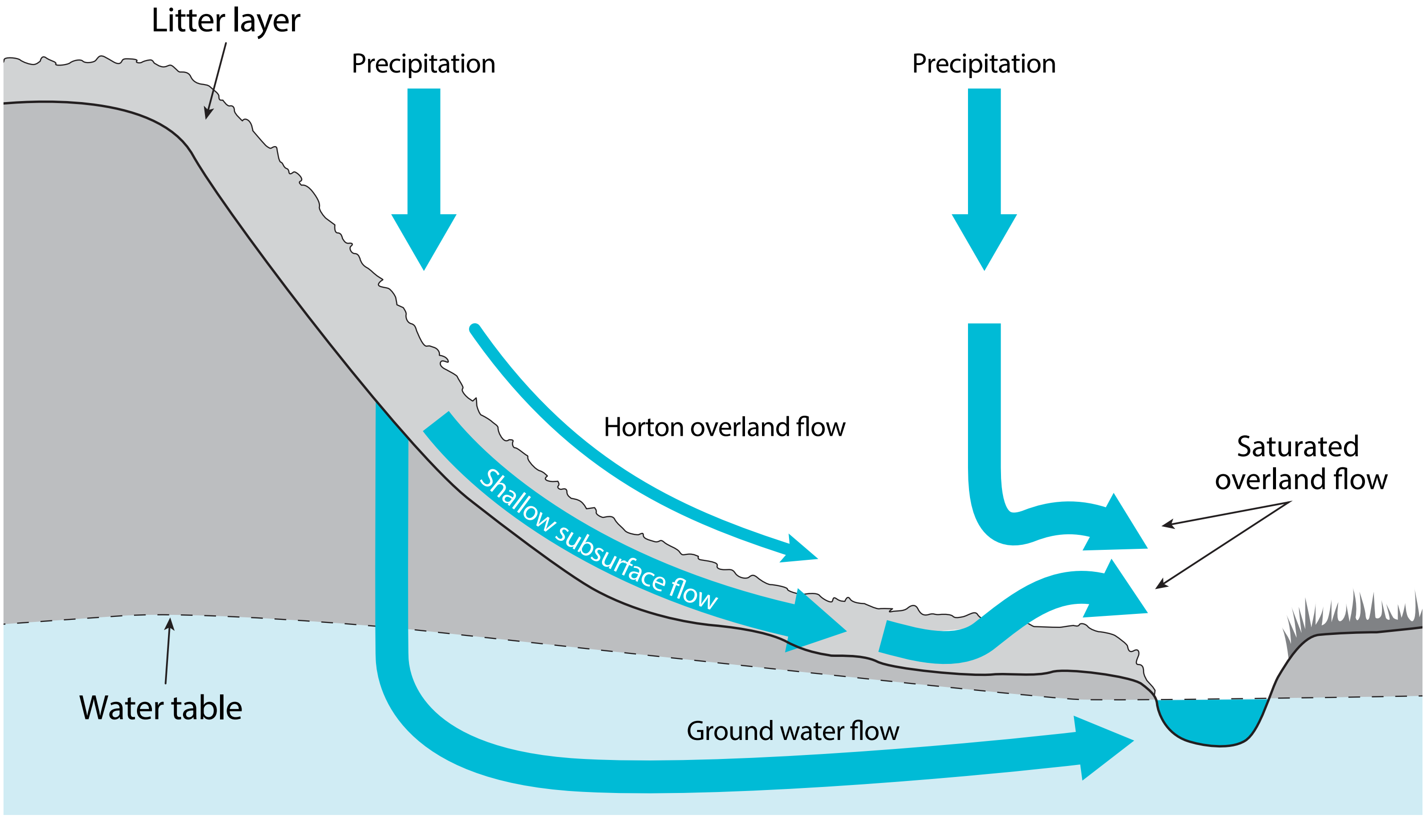


Figure 6.5. Water flow paths. Precipitation reaches a stream as surface runoff and by infiltration into the ground where it contributes to groundwater flow. The amount of water following each flow path depends on the capacity of the soil to absorb water, the amount of storage in surface depressions, and the amount, intensity, and duration of precipitation (image adapted from Stream Corridor Restoration Handbook; FISWRG, 1998).

bring in nutrients from upstream areas (Figure 6.5). As water moves downstream, it moves into and out of the hyporheic zone, carrying with it dissolved nutrients that react with the sediments and biota living in the interstitial spaces of the sediments (Fernald et al. 2006). Wondzell (2011) suggests that hyporheic exchange is likely to be most important in small streams but that hyporheic exchange through sand-bedded streams is substantial and may have an important influence on nutrient loads of large rivers. Regardless, large fractions of N are removed via denitrification in hyporheic sediments at all scales from headwater streams to large rivers (Peterson et al. 2001; Mulholland et al. 2004; Alexander et al. 2007).

Hydrology is the driving force for a range of physical and biogeochemical processes controlling retention and flux of nutrients entering the riparian ecosystem (Chapter 2). Nutrients may originate from neighboring areas via local flow systems throughout the riverscape, from regional groundwater flow systems, especially along reaches of the higher stream orders, or from a recirculation of river water in riparian areas during flooding (Dahl et al. 2007). Depending on flood intensity and frequency, water may overflow the streambanks and move into the floodplain where sediments may be deposited or, conversely, litter, woody material, and sediments may be mobilized and transported into the main channel.

Many floodplains in the Pacific Northwest are highly altered by human activities and are often artificially disconnected from stream channels by levees. Subsequently, natural flooding processes and nutrient exchange, both deposition and production, are severely altered often with negative consequences to riparian habitat.

Discharge volume, stream flow velocity, and temporal variability in flow, coupled with nutrient concentrations dictate the mass loading of each nutrient flowing downstream. Headwater streams commonly make up 75% or more of the total stream channel length in drainage basins (Benda et al. 2004) and have a disproportionately profound influence on the water quality and quantity for downstream ecosystems (Peterson et al. 2001). Alexander et al. (2007) found that fluxes from first-order headwater streams accounted for 70% of water volume and 65% of the N flux occurring from second-order streams. Research comparing perennial, intermittent, and ephemeral stream nutrient cycling processes indicates that flow intermittency can affect the concentrations, fluxes, and forms of nutrients moving from streams into downstream ecosystems (Von Schiller et al. 2011).

In western Oregon and Washington, floodplains experience extremes in water and sediment flux (Naiman et al. 2010). Floodplains often have saturated soils and shallow, upwelling groundwater. Such wet conditions influence N processing by creating anoxic, denitrifying conditions in organic soil horizons (Burt et al. 2002). However, many floodplains of the PNW are highly altered by human activities and are often artificially disconnected from the stream channel via levees constructed to convert lands to agriculture or other uses. Subsequently, nutrient exchange during flood events between aquatic and riparian ecosystems, via both mobilization and deposition of dissolved and suspended organic matter, is severely altered. River restoration often focuses on reconnecting floodplains and associated riparian areas to stream channels in order to restore natural hydrology and re-establish nutrient cycling functions (Bernhardt et al. 2005; Craig et al. 2008). The hydrologic interactions of uplands and riparian soils control nutrient cycling and transport (Mayer et al. 2007; Weitzman et al. 2014). Therefore, it is critical to maintain riparian vegetation and to control soil erosion and nutrient runoff from uplands (Mayer et al. 2007; Hoffman et al. 2009; Zhang et al. 2010; Sweeney and Newbold 2014).

Impacts to riparian ecosystems may include historic events not well documented and not easily recognized today. Such impacts may lead to erroneous understanding of baseline conditions or misidentification of pristine areas. For example, recent research suggests that legacy sediments, referring to sediments deposited in floodplains from historic upland erosion in response to deforestation and construction of milldams and ponds, have altered nutrient cycling in riparian ecosystems throughout the mid-Atlantic region of the US (Walter and Merritts 2008; Merritts et al. 2011). Much of the fine sediment carried by these streams during storms is from streambank erosion, which often includes legacy sediments, thereby contributing substantially to sediment and nutrient loads in streams (Gellis and Noe 2013). While milldams may not have been as common in the Pacific Northwest, splash dams created to move logs downstream were common throughout the region historically until the mid-20th century (Phelps 2011). These dams and other historic logging activities may have altered riparian systems in ways not easily recognized today but which may have had a significant legacy effect on sediment deposition in floodplains. For example, splash dam releases exceeded the effects of 100-year flood events in headwater regions and were comparable to 100-year flows in lower reaches (Phelps 2011). Splash dams also may have contributed to simplification of stream channels when rocks and debris jams were removed to facilitate log transport (Sedell and Luchessa 1981; Wohl 2000). Overall, the long-term effects of splash dams are not well documented nor are they accounted for in current riparian assessments.

South Fork Sauk River/Wendy Cole, WDFW

Precipitation and runoff are master variables regulating transport and transformation of C, N, and P in watersheds (Dosskey et al. 2010; Vidon et al. 2010). The interaction between land use and climate variability can increase the amplitude and frequency of nutrient pulses, effecting rapid and large changes in concentrations and fluxes of materials (Kaushal et al. 2014). Shifting patterns in runoff and temperature due to land use and climate variability can amplify pulses of greenhouse gases (GHG) such as CO_2, CH_4, and N_2O emanating from watersheds (Kaushal et al. 2014). For example, Vidon et al. (2014) showed strong N_2O pulses in response to storm events in a riparian wetland. Warming increases production of some GHGs in wetlands (Inglett et al. 2012) because temperature affects reaction kinetics and equilibria. Higher stream temperatures may increase rates of both bacterial consumption of nutrients and production of GHGs in streams and rivers (Kaushal et al. 2014). Recent research shows that groundwater may also warm in response to climate (Menberg et al. 2014).

Most N and P enters receiving waterbodies during storm events and during cooler seasons when temperatures are lower than optimal for microbial activity (Kaushal et al. 2010) or when plants are senescent. During drought, residence times may be long, leading to more denitrification in riparian soils, however N loads may be small overall because stream flow may be much lower than normal (Mayer et al. 2010b; Filoso and Palmer 2011). However, drought may also lower moisture levels in riparian soils, potentially reducing the occurrence of anoxic conditions required for denitrification (Groffman et al. 2003). Conversely, high flow conditions during “flashy” storm events may greatly compromise nutrient retention processes (Booth 2005) flushing far more solutes downstream including N and P (Cooper et al. 2014). Consequently, high flows are also responsible for the vast majority of particulate C, N, and P transport in most streams (Alexander et al. 2000; Moore and Wondzell 2005).

Stream channel geomorphology and geologic setting, especially stream channel width, depth, and sediment lithology strongly influence groundwater behavior. In turn, nutrient movement in riparian ecosystems is dependent upon groundwater, which can be highly variable and follow preferential flow paths. Striz and Mayer (2008) showed that groundwater movement may shift considerably depending on seasonal precipitation trends. Streams may be gaining, that is, receiving water from the uplands when precipitation is high, or losing surface water to the ground when soils are dry and drought conditions prevail. Water movement through deep groundwater flow paths back to the stream can take years or decades (Hinkle 2009) and in some cases there is very little exchange with regional deep groundwater (Vaccaro 2011).

Mass removal of nutrients and contaminants in streambed sediments is dependent upon residence times because microbial processes, such as denitrification, require time for complete reaction (Zarnetske et al. 2012). Nutrient concentrations in soils and water may vary on time scales across minutes and hours as plants or algae take up nutrients. Yet, seasonal and annual cycles are also evident based on flow and temperatures. Nutrients may also vary on spatial scales of millimeters to meters depending on plant distribution or litter deposition, and can also vary at regional and global scales depending on land use, geomorphology, and precipitation patterns.

A better understanding of the role hyporheic connections play in the processing of nutrients will improve our ability to design riparian management strategies that protect water quality.

6.3. State of Riparian Nutrient Dynamics Science

Recent advances in stream nutrient research reveal spatially and temporally variable processes and resulting pulses of nutrient transformation, storage, and release that can differ among locations and through time at the same locations. We also know that freshwater and riparian ecosystems vary tremendously across the PNW based on soils, climate, organisms, and human activity. Because most stream nutrient research originates from case studies, the application of case study results to novel conditions should be done cautiously.

Most studies of riparian nutrient dynamics have been conducted in forested watersheds; grassland and desert ecosystems in our region are less well studied but share some common processes and dynamics. Much of the information provided here demonstrates that riparian areas can have important effects on nutrient transfer from land to water, which subsequently have implications for ecosystem goods and services (e.g., water for drinking, swimming, fishing, boating, etc.) and habitat conditions for aquatic species.

Generally, if the goal of riparian management involves nutrients, management should include efforts to shorten nutrient spiraling lengths, increase nutrient retention, provide substrates to support riparian food webs, and maintain natural shading in order to minimize algal blooms. However, stimulating algal and plant growth is one way to shorten nutrient spiraling lengths by enhancing nutrient uptake, at least temporarily as nutrients will be returned after senescence. A longer-term goal of riparian management is to avoid excessive nutrient delivery to streams. Maintaining riparian vegetation, soil conditions, and flowpaths, etc. will fascilitate the removal of nutrients from surface and subsurface flow before entering the channel. If nutrient input is successfully controlled, light levels promoting algal blooms become much less of an issue.

The larvae of many caddisfly species use silk to make protective cases, which are often strengthened with gravel, sand, and twigs/ Ned Pittman, WDFW

Some of the critical gaps in our understanding occur where physical sampling for detecting water and material movement is most difficult. This occurs in the soil subsurface, the streambed, and hyporheic zones where determining flow dynamics requires multiple sampling wells (Striz and Mayer 2008) or surrogate measures such as stream temperature (Stonestrom and Constantz 2003). A better understanding of the role hyporheic connections play in the processing of nutrients will improve our ability to design riparian management strategies that protect water quality (Hinkle et al. 2001; Zarnetske et al. 2011, 2012), particularly by expanding these studies to better represent the range of conditions in Washington. However, the use of stable isotopes of N, P, C, and also O and H in water have helped to illuminate some of the more obscure processes like water movement into and out of trees (Brooks et al. 2010) and the uptake of N along entire stream reaches (Ashkenas et al. 2004). Satellite monitoring data (e.g. GRACE) and lidar imaging, which can provide high-resolution topographic data for inaccessible areas, are proving to be valuable tools for the study of nutrient dynamics at the watershed scale. Drone technology may eventually further provide local and current information on condition and environmental variables. Coupling these new data sources with the development of powerful computer models (e.g., VELMA) that predict watershed-scale nutrient movement (Abdelnour et al. 2011; Abdelnour et al. 2013) will greatly enhance our ability to extrapolate field level data to watershed scales and to test land use riparian management scenarios and climate change impacts on nutrient cycling behavior.

6.4. The Role of Management

Litter from riparian areas is clearly an important source of limiting nutrients and chemical energy for aquatic ecosystems. Inorganic nutrients, such as those in artificial fertilizers, do not mimic the foodweb role of litter because inorganic nutrients feed the autotrophic components of foodwebs, such as algae. Litter feeds heterotrophic components of foodwebs, such as benthic macroinvertebrates, many of which are prey for resident salmonids and juvenile salmon. Consequently, providing adequate amounts of litter to streams is an important issue for riparian area management. The results of Bilby and Heffner (2016) suggest that 95% of full litter delivery to streams can be achieved with buffer widths between 40 to 60% of site-potential tree height, depending on site conditions. Numerous factors dictate the transport and fate of nutrients through riparian systems. Managing riparian systems to limit negative impacts of nutrients on downstream resources is inherently complex. The role of management is largely to offset the impacts of human activities that increase nutrient loads and runoff, exacerbate the impacts of nutrients, and/or alter the fluxes of nutrients. Table 6.1 identifies impacts of human activities in the context of factors influencing riparian areas and describes practical management and restoration approaches that are known to mitigate nutrient impacts. The primary objective with respect to offsetting nutrient impacts among all categories of land use is to maintain existing riparian systems, restore degraded systems, and plant riparian buffers where they do not currently exist. Overall, management should focus on where in the watershed important nutrient dynamics that affect aquatic system integrity (or fish life) occur, and how to reduce or ameliorate various human disturbances that can affect nutrient dynamics at those places.

Although salmon runs and associated nutrient fluxes have declined over time in Oregon, Washington, Idaho, and California (Gresh et al. 2000), salmon could still play an important role in stream ecosystems by greatly increasing the quantity of salmon carcasses planted in streams or by greatly increasing harvest escapement (Gende et al. 2002). Addition of inorganic nutrients will not mimic the food web role of salmon in PNW watersheds because they do not directly mimic the complex organic matter from salmon carcasses supplied to heterotrophs such as aquatic insects and juvenile salmonids (Compton et al. 2006). Maintaining structures, such as large wood, and processes that retain nutrients are expected to allow greater utilization of salmon-derived nutrients in aquatic food webs.

Table 6.1. Major factors influencing nutrient dynamics in riparian ecosystems and the role of management in reducing impacts

Factors influencing riparian areas	Impacts of human activities	Role of management in reducing impacts
Land use - agriculture	Excess nutrient application and leaching into groundwater and surface water	Improve nutrient management to reduce nutrient loading - right time, right place, right source, right rate (4Rs)
	Compaction and vegetation damage from poor livestock management near streams	Protect and restore riparian areas to attenuate sediment and nutrient loads
	Riparian disturbance or lack of natural vegetation in riparian area	Protect and restore riparian areas to reduce light inputs that drive algal overproduction and eutrophication
Land use - urbanization	Impervious surfaces increase nutrients and sediment transfer to surface waters	Disconnect direct runoff from impervious surfaces to streams to control nutrient and sediment inputs
	Increased nutrient application via yard fertilizer and pet waste	Educate the public and incentivize behaviors to reduce nutrient loads
	Leaky and ineffective sewer and septic systems	Improve sewer systems, sewage treatment, storm drainage, and septic systems
Land use - forestry (harvest and road building)	Small increase in nutrient loadings	Seek alternatives to clear cutting and protect riparian areas
	Small to moderate increase in sediment transport	Seek alternatives to clear cutting; improve harvest practices, slope and soil management, and road construction; and protect riparian areas
Forest age and composition	Recent clear-cuts may lose the most nutrients	Extend harvest rotations because mature forests likely retain more nutrients than young forests
	Red Alder fixes large amounts of N, which remains in the system and contributes to productive soils of the Coast Range and western Cascades mountains	Recognize the role that Red Alder forests can play in stream nutrient levels
Climate and seasonality	Rapid movement of water during fall and winter rains, sometimes bypass riparian zones	Slow water movement across the landscape
	Rapid N cycling during the summer but low flows mean this inorganic nitrogen accumulates in the soil, fall rains displace this N and transport it to streams	Consider timing of fertilizer applications
	Spring season usually has peak algal blooms because nutrient supplies are high as is light availability before full leaf out.	
	Slower processing of nutrients in cold weather, although conifer trees take up nurients year round	
Elevation and topography	Steep slopes lead to higher rates of runoff and sediment transport	Minimize disturbance to vegetation and soils on steeper slopes through forest and pasture management

Table 6.1. CONTINUED

Major factors influencing nutrient dynamics in riparian ecosystems and the role of management in reducing impacts

Factors influencing riparian areas	Impacts of human activities	Role of management in reducing impacts
Hydrology	Reduced connection between the channel and floodplain can limit nutrient uptake	Reconnect flood plains to increase water residence times, which can increase ground water-surface interactions and nutrient uptake
Nutrient concentrations, forms, and inputs	High concentration and multiple sources of bioavailable nutrient forms may overload capacity of riparian systems to process nutrient loads	Consider terrestrial versus atmospheric sources; carefully manage synthetic fertilizer and manure applications, and balance N and P
Soil properties and geology	Low permeability reduces infiltration and interaction with soils	Practice good soil management because rich, fertile soils can contribute organic carbon for denitrification
	Sandy soils can transmit more water and nutrients (e.g., pumice soils and septic systems)	Recognize that some soils are better at adsorbing P (e.g., volcanic and highly weathered soils)
Biota	Declines in salmon runs reduces supply of nutrients and organic matter for invertebrates, birds, mammals, and other fish	Restore salmon populations, which increases nutrients in ecosystem: nutrients are assimilated by vegetation, consumed by animals, and converted by microbial processes

6.5. Conclusions

Riparian areas are important sources and sinks of organic matter and nutrients for streams. Numerous sources of C, N, and P exist, each taking various organic and inorganic forms, and each moving through the environment continuously along multiple paths at various temporal and spatial scales. Transfers may be via subsurface flow, direct input of materials via litterfall, treefall, or animal movement. Nutrient and energy subsidies from riparian areas to streams and rivers have profound effects on the composition, structure, and functions of aquatic ecosystems.

In watersheds where excess nutrients are polluting surface waters, hydrology and flow paths are critically important when considering potential for nutrient removal by riparian areas. If a riparian area is physically disconnected from the stream, flow paths may not carry materials through the riparian area, particularly into the hotspots with appropriate conditions for denitrification, and thus N removal will not occur as readily. Thus, maintaining hydrologic connections between riparian areas and stream channels are critical. This includes allowing natural seasonal patterns of flow including flooding.

Historically, many watersheds in Washington have seen substantial inputs of salmon-derived N, P, and organic matter. Riparian areas serve as the critical connection for transferring these nutrients to uplands via animal consumption, plant uptake, and biogeochemical cycling.

Results of a large review of riparian ecosystem studies across the globe indicated that vegetated riparian areas were more effective in trapping N than unvegetated ecosystems, but that the type of vegetation did not matter tremendously. Wider riparian areas were more effective in N removal across studies (see Chapter 5).

N-fixation by Red Alder is an important component of nutrient cycling in forests of western Washington and

Oregon. In the PNW, deciduous riparian forests tend to produce more litter with higher nutrient content and nutritional quality for stream food webs than conifer riparian areas. However, conifers have a longer lifespan, produce larger wood and are a more consistent source of shade. In riparian areas with a mixture of deciduous and conifer trees, deciduous species enhance litter flux and nutrient delivery to terrestrial and aquatic food webs, thus, complementing the provision of shade and large wood by conifers. Both the food web and structural roles of tree species should be considered in riparian management plans in order to support aquatic production.

Structurally diverse habitats with variability in geomorphology and soils are likely to support correspondingly diverse biota including vegetation, animals, and microbial communities that will be best able to process and assimilate nutrient loads. Headwaters are especially controlled by riparian processes and thus, are important zones of active nutrient processing that can significantly affect downstream transport of nutrients. Therefore, maintaining riparian areas in headwater systems may be disproportionately important to stream function. Similarly, maintaining hydrologic connection of streams with their floodplains is important because rich, organic sediments are deposited on floodplains during flood events. Conversely, channelization and stream incision from erosion disconnects streams from their floodplains, causes flashy flows, further incises stream channels while starving floodplains of regular resupply of organic matter, and can lower groundwater tables below the rooting zone thereby contributing to the reduction of riparian vegetation in arid climates.

While more research will be necessary to answer important questions about nutrients in riparian ecosystems, the science is very clear about the importance of the condition of riparian areas on the flows of nutrients through stream systems. Riparian areas and their streams are much more than conduits for nutrients from the uplands to downstream systems; they are vitally important locations for the short- and long-term supply, storage, and transformation of nutrients.

Perennial fruiting bodies, or conks, of a polypore fungi/Ned Pittman, WDFW

6.6. Literature Cited[2]

Abdelnour, A., M. Stieglitz, F. Pan, and R. McKane. 2011. Catchment hydrological responses to forest harvest amount and spatial pattern. Water Resources Research 47: W09521 (i).

Abdelnour, A., R. McKane, M. Stieglitz, F. Pan, and Y. Cheng. 2013. Effects of harvest on carbon and nitrogen dynamics in a Pacific Northwest forest catchment. Water Resources Research 49:1292-1313. (i)

Alexander, R.B., R. A. Smith, and G.E. Schwartz. 2000. Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico. Nature 403:758–761. (i)

Alexander, R.B., E.W. Boyer, R.A. Smith, G.E. Schwarz, and R.B. Moore. 2007. The role of headwater streams in downstream water quality. Journal of the American Water Resources Association 43:41-59. (i)

Ashkenas, L.R., S.L. Johnson, S.V. Gregory, J.L. Tank, and W.M. Wollheim. 2004. A stable isotope tracer study of nitrogen uptake and transformation in an old-growth forest stream. Ecology 85:1725-1739. (i)

Beechie, T., and T.H. Sibley. 1997. Relationships between channel characteristics, woody debris and fish habitat in northwestern Washington streams. Transactions of the American Fisheries Society 126:217-229. (i)

Benda, L., N.L. Poff, D. Miller, T. Dunne, G. Reeves, G. Pess, and M. Pollock. 2004. The network dynamics hypothesis: how channel networks structure riverine habitats. BioScience 54:413-427. (i)

Benda, L., M.A. Hassan, M. Church, and C.L. May. 2005. Geomorphology of steepland headwaters: the transition from hillslopes to channels. Journal of the American Water Resources Association 41:835-851. (i)

Ben-David, M., T.A. Hanley, and D.M. Schell. 1998. Fertilization of terrestrial vegetation by spawning Pacific salmon: the role of flooding and predator activity. Oikos 83:47-55. (i)

Benfield, E.F. 1997. Comparisons of litterfall input to streams. Journal of the North American Benthological Society 16:104-108. (i)

Bernhardt, E.S., M.A. Palmer, J.D. Allan, G. Alexander, K. Barnas, S. Brooks, J. Carr, S. Clayton, C. Dahm, J. Follstad-Shah, D. Galat, S. Gloss, P. Goodwin, D. Hart, B. Hassett, R. Jenkinson, S. Katz, G.M. Kondolf, P.S. Lake, R. Lave, J.L. Meyer, T.K. O'Donnell, L. Pagano, B. Powell, and E. Sudduth. 2005. Synthesizing U.S. river restoration efforts. Science 308:636-637. (i)

Bilby, R.E., R.R. Fransen, and P.A. Bisson. 1996. Incorporation of nitrogen and carbon from spawning Coho Salmon into the trophic system of small streams: evidence from stable isotopes. Canadian Journal of Fisheries and Aquatic Sciences 53:164–173. (i)

Bilby, R.E., and J.T. Heffner. 2016. Factors influencing litter delivery to streams. Forest Ecology and Management 369:29-37. (i)

Bisson, P.A., and R.E. Bilby. 1998. Organic matter and trophic dynamics. Page 373-398 *in* R.J. Naiman and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Bohn, H.L., B.L. McNeal, and G.A O'Connor. 1985. Soil chemistry. John Wiley and Sons, New York. (i)

Booth, D.B. 2005. Challenges and prospects for restoring urban streams: a perspective from the Pacific Northwest of North America. Journal of North American Benthological Society 24:724–737. (i)

Brooks, J.R., H. Barnard, R. Coulombe, and J.J. McDonnell. 2010. Ecohydrologic separation of water between trees and streams in a Mediterranean climate. Nature Geoscience 3:100-104. (i)

Brown G.W., A.R. Gahler, and R.B Marston. 1973. Nutrient losses after clear-cut logging and slash burning in the Oregon Coast Range. Water Resources Research 9:1450-1453. (i)

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[2] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Burt, T.P., G. Pinay, F.E. Matheson, N.E. Haycock, A. Butturini, J.C. Clement, S. Danielescu, D.J. Dowrick, M.M. Hefting, A. Hillbricht-Ilkowska, and V. Maitre. 2002. Water table fluctuations in the riparian zone: comparative results from a pan-European experiment. Journal of Hydrology 265:129–148. (i)

Cardinale, B.J., M.A. Palmer, C.M. Swan, S. Brooks, and N.L. Poff. 2002. The influence of substrate heterogeneity on biofilm metabolism in a stream ecosystem. Ecology 83:412–422. (i)

Cardinale, B.J. 2011. Biodiversity improves water quality through niche partitioning. Nature 472:86-89. (i)

Carpenter, S., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V.H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Issues in Ecology No. 3, Ecological Society of America, Washington, D.C. (i)

Cederholm, C.J., D.B. Houston, D.I. Cole, and W.J. Scarlett. 1989. Fate of Coho Salmon *Oncorhynchus kisutch* carcasses in spawning streams. Canadian Journal of Fisheries and Aquatic Sciences 46:1347–55. (i)

Cederholm, C.J., M.D. Kunze, T. Murota, and A. Sibatani. 1999. Pacific salmon carcasses: essential contributions of nutrients and energy for aquatic and terrestrial ecosystems. Fisheries 24:6–15. (i)

Chappell, H.N., D.W. Cole, S.P. Gessel, and R.B. Walker. 1991. Forest fertilization research and practice in the Pacific Northwest. Fertilizer Research 27:129-140. (i)

Compton, J.E., M.R. Church, S.T. Larned, and W.E. Hogsett. 2003. Nitrogen export from forested watersheds of the Oregon Coast Range: the role of N_2-fixing Red Alder. Ecosystems 6:773–85. (i)

Compton, J.E., and D.W. Cole. 1998. Phosphorus cycling and soil P fractions in Douglas-fir and Red Alder stands. Forest Ecology and Management 110:101-112. (i)

Compton, J.E., C.P. Andersen, D.L. Phillips, J.R. Brooks, M.G. Johnson, M.R. Church, W.E. Hogsett, M.A. Cairns, P.T. Rygiewicz, B.C. McComb, and C.D. Shaff. 2006. Ecological and water quality consequences of nutrient addition for salmon restoration in the Pacific Northwest. Frontiers in Ecology and the Environment 4:18–26. (i)

Cooper, C.A., P.M. Mayer, and B.R. Faulkner. 2014. Effects of road salts on groundwater and surface water dynamics of sodium and chloride in an urban restored stream. Biogeochemistry 121:149-166. (i)

Craig, L.S., M.A. Palmer, D.C Richardson, S. Filoso, E.S. Bernhardt, B.P. Bledsoe, M.W. Doyle, P.M. Groffman, B.A. Hassett, S.S. Kaushal, P.M. Mayer, S.M. Smith, and P.R. Wilcock. 2008. Stream restoration strategies for reducing river nitrogen loads. Frontiers in Ecology and the Environment 6:529-538. (i)

Dahl, M., B. Nilsson, J.H. Langhoff, and J.C. Refsgaard. 2007. Review of classification systems and new multi-scale typology of groundwater–surface water interaction. Journal of Hydrology 344:1–16. (i)

Denver, J.M., C.A. Cravotta III, S.W. Altor, and B.D. Lindsey. 2010. Contributions of phosphorus from groundwater to streams in the Piedmont, Blue Ridge, and Valley and Ridge Physiographic Provinces, Eastern United States. Scientific Investigations Report 2010–5176. U.S. Geological Survey, Reston, Virginia. (viii)

Domagalski J.L., and H. Johnson. 2012. Phosphorus and groundwater: establishing links between agricultural use and transport to streams. U.S. Geological Survey, California Water Science Center, Sacramento. (viii)

Dosskey, M.G., P. Vidon, N.P. Gurwick, C.J. Allan, T.P. Duval, and R. Lowrance. 2010. The role of riparian vegetation in protecting and improving chemical water quality in streams. Journal of the American Water Resources Association 46:261-277. (i)

Dudley-Williams, D., and N.E. Williams. 1993. The upstream/downstream movement paradox of lotic invertebrates: quantitative evidence from a Welsh mountain stream. Freshwater Biology 30:199-218. (i)

Edmonds, R.L., T.B. Thomas, and R.D. Blew. 1995. Biogeochemistry of an old-growth forested watershed, Olympic National Park, Washington. Water Resources Bulletin 31:409-419. (i)

FISRWG (Federal Interagency Stream Restoration Working Group). 1998. Stream corridor restoration: principles, processes, and practices. NEH-653. Natural Resources Conservation Service, Washington, D.C. (viii)

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management: an ecological, economic and social assessment. U.S. Department of Agriculture and U.S. Department of the Interior, Portland, Oregon. (viii)

Fernald, A.G., D.H. Landers, and P.J. Wigington. 2006. Water quality changes in hyporheic flow paths between a large gravel bed river and off-channel alcoves in Oregon, USA. River Research and Applications 22:1111-24. (i)

Filoso, S., and M.A. Palmer. 2011. Assessing stream restoration effectiveness at reducing nitrogen export to downstream waters. Ecological Applications 21:1989–2006. (i)

Gellis, A.C., and G.B. Noe. 2013. Sediment source analysis in Linganore Creek watershed, Maryland, USA, using the sediment fingerprinting approach: 2008 to 2010. Journal of Soils and Sediments 13:1735–1753. (i)

Gende, S.M., R.T. Edwards, M.F. Willson, and M.S. Wipfli. 2002. Pacific salmon in aquatic and terrestrial ecosystems. BioScience 52:917-28. (i)

Giardina, C., S. Huffman, D. Binkley, and B. Caldwell. 1995. Alders increase phosphorus supply in a Douglas-fir plantation. Canadian Journal of Forest Research 25:1652-1657. (i)

Giesler, R., T. Andersson, L. Lövgren, and P. Persson. 2005. Phosphate sorption in aluminum- and iron-rich humus soils. Soil Science Society of America 69:77–86. (i)

Greathouse, E.A., J.E. Compton, and J. Van Sickle. 2014. Linking landscape characteristics and high stream nitrogen in the Oregon Coast Range: Red Alder complicates use of nutrient criteria. Journal of the American Water Resources Association 50:1383-1400. (i)

Gresh, T., J. Lichatowich, and P. Schoonmaker. 2000. An estimation of historic and current levels of salmon production in the Northeast Pacific Ecosystem: evidence of a nutrient deficit in the freshwater systems of the Pacific Northwest. Fisheries 25:15-21. (i)

Groffman, P.M., N.J. Boulware, W.C. Zipperer, R.V. Pouyat, L.E. Band, and M.F. Colosimo. 2002. Soil nitrogen cycling processes in urban riparian zones. Environmental Science & Technology 36:4547-4552. (i)

Groffman, P.M., D.J. Bain, L.E. Band, K.T. Belt, G.S. Brush, J.M. Grove, R.V. Pouyat, I.C. Yesilonis, and W.C. Zipperer. 2003. Down by the riverside: urban riparian ecology. Frontiers in Ecology and the Environment 1:315–321. (i)

Groffman, P.M., A.M. Dorsey, and P.M. Mayer. 2005. Nitrogen processing within geomorphic features in urban streams. Journal of North American Benthological Society 24:613-625. (i)

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack, Jr., and K.W. Cummins, 1986. Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research 15:133-302. (i)

Hart, S.K., D.E. Hibbs and S.S. Perakis. 2013. Riparian litter inputs to streams in the central Oregon Coast Range. Freshwater Science 32:343-358. (i)

Hedin, L.O., J.C. von Fischer, N.E. Ostrom, B.P. Kennedy, M.G. Brownand, and G.P. Robertson. 1998. Thermodynamic constraints on nitrogen transformations and other biogeochemical processes at soil-stream interfaces. Ecology 79:684-703. (i)

Hefting, M., J.C. Clement, D. Dowrick, A.C. Cosandey, S. Bernal, C. Cimpian, A. Tatur, T. P. Burt, and G. Pinay. 2004. Water table elevation controls on soil nitrogen cycling in riparian wetlands along a European climatic gradient. Biogeochemistry 67:113-134. (i)

Helfield J.M., and R. Naiman. 2001. Effects of salmon-derived nitrogen on riparian forest growth and implications for stream productivity. Ecology 82:2403-2409. (i)

Hershey, A.E., J. Pastor, B.J. Peterson, and G.W. Kling. 1993. Stable isotopes resolve the drift paradox for Baetis mayflies in an arctic river. Ecology 74:2315-2325. (i)

Hinkle, S.R., J.H. Duff, F.J. Triska, A. Laenen, E.B. Gates, K.E. Bencala, D.A. Wentz, and S.R. Silva. 2001. Linking hyporheic flow and nitrogen cycling near the Willamette River—a large river in Oregon, USA. Journal of Hydrology 244:157-80. (i)

Hinkle, S.R. 2009. Tritium/helium-3 apparent ages of shallow ground water, Portland basin, Oregon, 1997–98: Scientific Investigations Report 2009-5057. U.S. Geological Survey, Reston, Virginia. (viii)

Hoffmann, C.C., C. Kjaergaard, J. Uusi-Kämppä, H.C. Bruun, and B. Kronvang. 2009. Phosphorus retention in riparian buffers: review of their efficiency. Journal of Environmental Quality 38:1942–1955. (i)

Ice, G., and D. Binkley. 2003. Forest streamwater concentrations of nitrogen and phosphorus: comparison with EPA's proposed water quality criteria. Journal of Forestry 101:21–28. (i)

Inglett K.S., P.W. Inglett, K.R. Reddy, and T.Z. Osborne. 2012. Temperature sensitivity of greenhouse gas production in wetland soils of different vegetation. Biogeochemistry 108:77-90. (i)

Jacoby J.M., and J. Kann. 2007. The occurrence and response to toxic cyanobacteria in the Pacific Northwest, North America. Lake Reservoir Management 23:123–143. (i)

Johnson, D.W., and D.W. Cole. 1980. Anion mobility in soils: relevance to nutrient transport from forest ecosystems. Environment International 3:79-90. (i)

Jordan, S.J., J. Stoffer, and J.A. Nestlerode. 2011. Wetlands as sinks for reactive nitrogen at continental and global scales: a meta-analysis. Ecosystems 14:144-155. (i)

Kaushal, S.S., M. Pace, P. Groffman, L. Band, K. Belt, P. Mayer, and C. Welty. 2010. Land use and climate variability amplify contaminant pulses. EOS Transactions of the American Geophysical Union 91:221-222. (i)

Kaushal, S.S., and W.M. Lewis Jr. 2005. Fate and transport of dissolved organic nitrogen in minimally disturbed streams of Colorado, USA. Biogeochemistry 74:303–321. (i)

Kaushal, S.S., P.M. Mayer, P.G. Vidon, R.M. Smith, M.J. Pennino, S. Duan, T.A. Newcomer, C. Welty, and K. Belt. 2014. Land use and climate variability amplify carbon, nutrient, and contaminant pulses: a review with management implications. Journal of the American Water Resources Association 50:585-614. (i)

Lazar, J.G., A.J. Gold, K. Addy, P.M. Mayer, K.J. Forshay, and P.M. Groffman. 2014. Instream large wood: denitrification hotspots with low N_2O production. Journal of the American Water Resources Association 50:615-625.

Martin, C.W., and R.D. Harr. 1989. Logging of mature Douglas-fir in western Oregon has little effect on nutrient output budgets. Canadian Journal of Forest Research 19:35-43. (i)

Masese, F.O., K.G. Abrantes, G.M. Gettel, S. Bouillon, K. Irvine, and M.E. McClain. 2015. Are large herbivores vectors of terrestrial subsidies for riverine food webs? Ecosystems 18:686-706. (i)

Mayer, P.M., A.H. Todd, J.A. Okay, and K.A. Dwire. 2010a. Introduction to the featured collection on riparian ecosystems and buffers. Journal of the American Water Resources Association 46:207-210. (i)

Mayer, P.M., P.M. Groffman, E. Striz, and S.S. Kaushal. 2010b. Nitrogen dynamics at the groundwater-surface water interface of a degraded urban stream. Journal of Environmental Quality 39:810–823. (i)

Mayer, P.M., K.A. Dwire, J.A. Okay, and P. Vidon. 2014. Introduction to the featured collection on riparian ecosystems. Journal of the American Water Resources Association 50:529-532. (i)

Mayer, P.M., S.K. Reynolds, M.D. McMutchen, and T.J. Canfield. 2007. Meta-analysis of nitrogen removal in riparian buffers. Journal of Environmental Quality 36:1172-1180. (i)

McClain, M.E., R.E. Bilby, and F.J. Triska. 1998. Biogeochemistry of N, P, and S in Northwest rivers: natural distributions and responses to disturbance. Pages 347-372 *in* R.J Naiman and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

McClain, M.E., E.W. Boyer, C.L. Dent, S.E. Gergel, N.B. Grimm, P.M. Groffman, S.C. Hart, J.W. Harvey, C.A. Johnston, E. Mayorga, W.H. McDowel,l and G. Pinay. 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6:301–312. (i)

McDowell, W.H., and G.E. Likens. 1988. Origin, composition, and flux of dissolved organic carbon in the Hubbard Brook Valley. Ecological Monographs 58:177-195. (i)

Menberg, K., P. Blum, B.L. Kurylyk, and P. Bayer. 2014. Observed groundwater temperature response to recent climate change. Hydrology and Earth System Sciences 18:4453–4466. (i)

Merritts, D., R. Walter, M. Rahnis, J. Hartranft, S. Cox, A. Gellis, N. Potter, W. Hilgartner, M. Langland, L. Manion, C. Lippincott, S. Siddiqui, Z. Rehman, C. Scheid, L. Kratz, A. Shilling, M. Jenschke, K. Datin, E. Cranmer, A. Reed, D. Matuszewski, M. Voli, E. Ohlson, A. Neugebauer, A. Ahamed, C. Neal, A. Winter, and S. Becker S. 2011. Anthropocene streams and base-level controls from historic dams in the unglaciated mid-Atlantic region, USA. Philosophical Transactions of the Royal Society A 369:976–1009. (i)

Moore, R.D., and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: a review. Journal of the American Water Resources Association 41:763-784. (i)

Moyle, P., A. Engilis, M. Escobar, C.M. Mosser, D. Purkey, L.C., and M.L. Truan. 2009. Progress report: innovative management options to prevent loss of ecosystem services provided by Chinook Salmon in California: overcoming the effects of climate change. US EPA NCER program. Available: http://cfpub.epa.gov/ncer_abstracts/index.cfm/fuseaction/display.abstractDetail/abstract/9070/-report/2009. (January 2019). (viii).

Mulholland, P.J., E.R. Marzolf, S.P. Hendricks, R.V. Wilkerson, and A.K. Baybayan. 1995. Longitudinal patterns of nutrient cycling and periphyton characteristics in streams: a test of upstream-downstream linkage. Journal of North American Benthological Society 14:357-370. (i)

Mulholland, P.J., A.M. Helton, G.C. Poole, R.O. Hall Jr., S.K. Hamilton, B.J. Peterson, J.L. Tank, L.R. Ashkenas, L.W. Cooper, C.N. Dahm, W.K. Dodds, S.E.G. Findlay, S.V. Gregory, N.B. Grimm, S.L. Johnson, W.H. McDowell, J.L. Meyer, H.M. Valett, J.R. Webster, C.P. Arango, J.J. Beaulieu, M.J. Bernot, A.J. Burgin, C.L. Crenshaw, L.T. Johnson, B.R. Niederlehner, J.M. O'Brien, J.D. Potter, R.W. Sheibley, D.J. Sobota, and S.M. Thomas. 2008. Stream denitrification across biomes and its response to anthropogenic nitrate loading. Nature 452:202-206. (i)

Murphy, M.L. 1998. Primary productivity. Pages 144-168 *in* R.J. Naiman and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York (i)

Naiman, R.J., J.S. Bechtold, T.J. Beechie, J.J. Latterell, and R. Van Pelt. 2010. A process-based view of floodplain forest patterns in coastal river valleys of the Pacific Northwest. Ecosystems 13:1-31. (i)

Naiman, R.J., and K.H. Rogers. 1997. Large animals and system level characteristics in river corridors. BioScience 47:521–9. (i)

Naiman, R.J., C.A. Johnston, and J.C. Kelley. 1988. Alteration of North American streams by Beaver. BioScience 38:753-762. (i)

Naiman, R.J., G. Pinay, C.A. Johnson, and J. Pastor. 1994. Beaver-induced influences on the long-term biogeochemical characteristics of boreal forest drainage networks. Ecology 75:905-921. (i)

Naiman, R.J., R.E. Bilby, D.E. Schindler, and J.M. Helfield. 2002. Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. Ecosystems 5:399-417. (i)

Newbold, J.D., R.V. O'Neill, J.W. Elwood, and W. Van Winkle. 1982. Nutrient spiralling in streams: implications for nutrient limitation and invertebrate activity. The American Naturalist 120:628-652. (i)

Newcomer, T.A., S.S. Kaushal, P.M. Mayer, A.R. Shields, E.A. Canuel, P.M. Groffman, and A.J. Gold. 2012. Influence of natural and novel organic carbon sources on denitrification in forested, degraded-urban, and restored streams. Ecological Monographs 82:449–466. (i)

Perakis, S.S., D.A. Maguire, T.D. Bullen, K. Cromack, R.H. Waring, and J.R. Boyle. 2006. Coupled nitrogen and calcium cycles in forests of the Oregon Coast Range. Ecosystems 9:63-74. (i)

Peterson, B.J., W.M. Wollheim, P.J. Mulholland, J.R. Webster, J.L. Meyer, J.L. Tank, E. Marti, W.B. Bowden, H.M. Valett, A.E. Hershey, W.H. McDowell, W.K. Dodds, S.K. Hamilton, S. Gregory, and D.D. Morrall. 2001. Control of nitrogen export from watersheds by headwater streams. Science 292:86-90. (i)

Peterson, C.E., and J.W. Hazard. 1990. Regional variation in growth response of coastal Douglas-fir to nitrogen fertilizer in the Pacific Northwest. Forest Science 36:625–640. (i)

Phelps, J.D. 2011. The geomorphic legacy of splash dams in the southern Oregon Coast Range. Master's thesis. University of Oregon. Eugene. (i)

Pingram M.A., K.J. Collier, D.P. Hamilton, B.O. David, and B.J. Hicks. 2012. Carbon sources supporting large river food webs: a review of ecological theories and evidence from stable isotopes. Freshwater Reviews 5:85–103. (i)

Pollock, M.M., T.J. Beechie, and C.E. Jordan. 2007. Geomorphic changes upstream of Beaver dams in Bridge Creek, an incised stream channel in the interior Columbia River basin, eastern Oregon. Earth Surface Processes and Landforms 32:1174-1185. (i)

Reddy K.R., O.A. Diaz, L.J. Scinto, and M. Agami. 1995. Phosphorus dynamics in selected wetlands and streams of the Lake Okeechobee basin. Ecological Engineering 5:183–207. (i)

Richardson, C.J., and P.E. Marshall. 1986. Processes controlling movement, storage and export of phosphorus in a fen peatland. Ecological Monographs 56:279–302. (i)

Richardson, J.S., and R.J. Danehy. 2007. A synthesis of the ecology of headwater streams and their riparian zones in temperate forests. Forest Science 53:131-147. (i)

Royer, T.V., and M.B. David. 2005. Export of dissolved organic carbon from agricultural streams in Illinois, USA. Aquatic Sciences 67:465–471. (i)

Sabater, S., H. Guasch, A. Roman, and I. Munoz. 2002. The effect of biological factors on the efficiency of river biofilms in improving water quality. Hydrobiologia 469:149-156. (i)

Sabater S., S.V. Gregory, and J.R. Sedell. 1998. Community dynamics and metabolism of benthic algae colonizing wood and rock substrata in forest stream. Journal of Phycology 34:561-567. (i)

Schechter, S.P., T.J. Canfield, and P.M. Mayer. 2013. A meta-analysis of phosphorus attenuation in best management practices (BMP) and low impact development (LID) practices in urban and agricultural areas. EPA 600/R-13/208. National Risk Management Research Laboratory, Ada, Oklahoma. (viii)

Scrivener, J.C. 1975. Water, water chemistry, and hydrochemical balance of dissolved ions in Carnation Creek watershed, Vancouver Island, July 1971 - May 1974. Technical Report 564. Canadian Fisheries and Marine Service, Vancouver, British Columbia. (viii)

Sedell, J.R., and K.J. Luchessa. 1981. Using the historical record as an aid to salmonid enhancement. Pages 210-223 *in* N.B. Armantrout, editor. Acquisition and utilization of aquatic habitat inventory information. Western Division, American Fisheries Society, Portland, Oregon. (viii)

Shen, J., L. Yuan, J. Zhang, H. Li, Z. Bai, X. Chen, W. Zhang, and F. Zhang. 2011. Phosphorus dynamics: from soil to plant. Plant Physiology 156:997-1005. (i)

Silvester, W.B. 1989. Molybdenum limitation of asymbiotic nitrogen fixation in forests of Pacific Northwest America. Soil Biology and Biochemistry 21:283-289. (i)

Sobczak, W.V., S.E.G. Findlay, and S. Dye. 2003. Relationships between DOC bioavailability and nitrate removal in an upland stream: an experimental approach. Biogeochemistry 62:309–327. (i)

Sobota, D.J., S. Johnson, S.V. Gregory, and L.R. Ashkenas, LR. 2012. A stable isotope tracer study of the influences of adjacent land use and riparian condition on fates of nitrate in streams. Ecosystems 15:1-17. (i)

Sollins, P., C.C. Grier, F.M. McCorison, K. Cromack Jr., R. Fogel, and R.L. Fredriksen. 1980. The internal element cycles of an old-growth Douglas-fir ecosystem in western Oregon. Ecological Monographs 50:261-285. (i)

Stanley, E.H., S.M. Powers, N.R. Lottig, I. Buffam, and J.T. Crawford. 2012. Contemporary changes in dissolved organic carbon (DOC) in human-dominated rivers: is there a role for DOC management? Freshwater Biology 57:26-42. (i)

Stonestrom, D.A., and J. Constantz. 2003. Heat as a tool for studying the movement of groundwater near streams. Circular 1260. U.S. Geological Survey, Denver, Colorado. (viii)

Striz, E.A., and P.M. Mayer. 2008. Assessment of near-stream groundwater-surface water interaction (GSI) of a degraded stream before restoration. EPA/600/R-07/058, National Risk Management Research Laboratory, Ada, Oklahoma. (viii)

Swain, N.R., and J.D. Reynolds. 2015. Effects of salmon-derived nutrients and habitat characteristics on population densities of stream-resident sculpins. PLOS ONE 10(6): e0116090. (i)

Sweeney, B.W., and J.D. Newbold. 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: a literature review. Journal of the American Water Resources Association 50:560-584. (i)

Sweeney, B.W., T.L. Bott, J.K. Jackson, L.A. Kaplan, J.D. Newbold, L.J. Standley, W.C. Hession, and R.J. Horwitz. 2004. Riparian deforestation, stream narrowing, and loss of stream ecosystem services. Proceedings of the National Academy of Sciences 101:14132-14137. (i)

Tague, C., and G.E. Grant. 2004. A geological framework for interpreting the low-flow regimes of Cascade streams, Willamette River Basin, Oregon. Water Resources Research 40:W04303. (i)

Tarrant, R.F., L.A. Issac, and R.F. Chandler. 1951. Observations on litter fall and foliage nutrient content of some Pacific Northwest tree species. Journal of Forestry 49:914-915. (i)

Triska, F.J., J.R. Sedell, K. Cromack Jr., S.V. Gregory, and F.M. McCorison. 1984. Nitrogen budget for a small coniferous forest stream. Ecological Monographs 54:119-40. (i)

USEPA ATTAINS database. Available: http://www2.epa.gov/nutrient-policy-data/waters-assessed-impaired-due-nutrient-related-causes. (June 2015). (vi)

Vaccaro, J.J. 2011. River-aquifer exchanges in the Yakima River basin, Washington: U.S. Geological Survey Scientific Investigations Report 2011-5026. U.S. Geological Survey, Reston, Virginia. (viii)

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fish and Aquatic Science 37:130-137. (i)

Vidon, P., C. Allan, D. Burns, T.P. Duval, N. Gurwick, S. Inamdar, R. Lowrance, J. Okay, D. Scott, and S. Sebestyen. 2010. Hot spots and hot moments in riparian zones: potential for improved water quality. Journal of the American Water Resources Association 46:278-298. (i)

Vidon, P., P.A. Jacinthe, X. Liu, K. Fisher, and M. Baker. 2014. Hydrobiogeochemical controls on riparian nutrient and greenhouse gas dynamics: 10 Years post-restoration. Journal of the American Water Resources Association 50:639-652. (i)

Von Schiller, D., V. Acuña, D. Graeber, E. Martí, M. Ribot, S. Sabater, X. Timoner, and K. Tockner. 2011. Contraction, fragmentation and expansion dynamics determine nutrient availability in a Mediterranean forest stream. Aquatic Sciences 73:485-497. (i)

Wallace, J.B., S.L. Eggert, J.L. Meyer, and J.R. Webster. 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. Science 277:102-104. (i)

Walter, R.C., and D.J. Merritts. 2008. Natural streams and the legacy of water-powered mills. Science 319:299–304. (i)

Weitzman, J.N., K.J. Forshay, J.P. Kaye, P.M. Mayer, J. Koval, and R.C. Walter. 2014. Potential nitrogen and carbon processing in a landscape rich in mill-dam legacy sediments. Biogeochemistry 120:337-357. (i)

Welch, E.B., J.M. Jacoby and C.W. May. 1998. Stream quality. Pages 69-94 *in* R.J. Naiman, and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Wigington, P.J. Jr., T.J. Moser, and D.R. Lindeman. 2005. Stream network expansion: a riparian water quality factor. Hydrological Processes 19:1715–1721. (i)

Wigington, P.J., M.R. Church, T.C. Strickland, K.N. Eshleman, and J. Van Sickle. 1998. Autumn chemistry of Oregon Coast Range streams. Journal of the American Water Resources Association 34:1035-49. (i)

Willson, M.F., and K.C. Halupka. 1995. Anadromous fish as keystone species in vertebrate communities. Conservation Biology 9:489–97. (i)

Wilzbach, M.A., B.C. Harvey, J.L. White, and R.J. Nakamoto. 2005. Effects of riparian canopy opening and salmon carcass addition on the abundance and growth of resident salmonids. Canadian Journal of Fisheries and Aquatic Sciences 62:58–67. (i)

Wipfli, M.S. 2005. Trophic linkages between headwater forests and downstream fish habitats: implications for forest and fish management. Landscape and Urban Planning 72:205-213. (i)

Wipfli, M.S., and C.V. Baxter. 2010. Linking ecosystems, food webs, and fish production: subsidies in salmonid watersheds. Fisheries 35:373-387. (i)

Wipfli, M.S., and J. Musslewhite. 2004. Density of Red Alder (*Alnus rubra*) in headwaters influences invertebrate and detritus subsidies to downstream fish habitats in Alaska. Hydrobiologia 520:153-163.

Wise, D.R., and H.M. Johnson. 2013. Application of the SPARROW model to assess surface-water nutrient conditions and sources in the United States Pacific Northwest. Scientific Investigations Report 2013–5103. U.S. Geological Survey, Reston, Virginia. Available: http://pubs.usgs.gov/sir/2013/5103/. (Januray 2019) (viii)

Wise, D.R., and H.M. Johnson. 2011. Surface-water nutrient conditions and sources in the United States Pacific Northwest. Journal of the American Water Resources Association 47:1110-1135. (i)

Wohl, E.E. 2000. Mountain Rivers. The American Geophysical Union, Washington, D.C. (viii)

Wondzell, S.M. 2011. The role of the hyporheic zone across stream networks. Hydrological Processes 25:3525-3532. (i)

Xiong, S., and C. Nilsson. 1997. Dynamics of leaf litter accumulation and its effects on riparian vegetation: a review. Botanical Review 63:240–264. (i)

Yanai, R.D. 1992. Phosphorus budget of a 70-yr-old northern hardwood forest. Biogeochemistry 17:1-22. (i)

Zarnetske, J.P., R. Haggerty, S.M. Wondzell, V.A. Bokil, and R. González-Pinzón. 2012. Coupled transport and reaction kinetics control the nitrate source-sink function of hyporheic zones. Water Resources Research 48: W11508. (i)

Zarnetske, J.P., R. Haggerty, S.M. Wondzell, and M.A. Baker. 2011. Dynamics of nitrate production and removal as a function of residence time in the hyporheic zone. Journal of Geophysical Research 116:G01025. (i)

Zhang, X., X. Liu, M. Zhang, R.A. Dahlgren, and M. Eitzel. 2010. A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. Journal of Environmental Quality 39:76-84. (i)

Zou, X., D. Binkley, and B.A. Caldwell. 1995. Effects of dinitrogen-fixing trees on phosphorus biogeochemical cycling in contrasting forests. Soil Science Society of America Journal 59:1452-1458. (i)

Big Creek Trail/Rachel Blomker, WDFW

Chapter 7. Riparian Areas of the Columbia Plateau

By: George F. Wilhere

This chapter is dedicated Karin Divens who passed away unexpectedly on 12 October 2017. Karin made a significant contribution to this chapter by organizing a two-day, multi-disciplinary workshop that advanced our scientific understanding of riparian areas in the Columbia Plateau Ecoregion. Karin was an effective and well-respected habitat biologist at WDFW, who brought not only science to her work but a full heart.

7.1. Introduction

The Columbia Plateau Ecoregion (sensu Omernick 1987),[1] is classified as a cold desert (Sleeter 2012), and covers about one-third of Washington State (Figure 7.1). In the driest parts of the ecoregion, a rain shadow cast by the Cascades Mountains limits precipitation to 6 to 9 in (15 to 23 cm) per year, and even wettest parts of the ecoregion, the Palouse Hills Subregion, receive only 18 to 23 in (46 to 58 cm) per year (Bryce and Omernik 1997). Uplands of the Columbia Plateau are covered by either shrub-steppe or steppe[2] vegetation (i.e., sagebrush and bunchgrass communities, respectively), and, most notably, lack trees (Franklin and Dyrness 1988). The ecoregion's desert climate confines nearly all trees to riparian areas.[3]

Seventy-five years ago Daubenmire (1942) wrote, "The vegetation of the unforested regions in the Pacific Northwest has been less thoroughly studied and is consequently less perfectly understood than is that of the forested areas." The situation has not changed. Most of what is known about riparian areas in the Pacific Northwest is based on research conducted in forested ecoregions. During my review of the scientific literature, I found far less information about the composition, structure, and functions of riparian ecosystems in unforested ecoregions, such as the Columbia Plateau, than I found for forested ecoregions. However, many of the basic principles and qualitative relationships that have emerged from research in forested ecoregions are also valid in unforested ecoregions. For instance, while there are significant quantitative differences among ecoregions in the amount of wood or shade provided by riparian areas, the physical processes of large wood recruitment and stream shading should be very nearly the same. On the other hand, processes that are strongly affected by groundwater, such as nutrient dynamics and pollutant removal, should be different. To clarify current

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[1] There are several different delineations of ecoregions (e.g., USFWS 1994; Bailey 1995; Olson et al. 2001). For this report WDFW used the ecoregions of Omernick (1987). The only non-forested ecoregion in Washington is the Columbia Plateau. Forested ecoregions surrounding the Columbia Plateau in Washington are the Eastern Cascades, North Cascades, Okanogan, Northern Rockies, and Blue Mountains.

[2] Steppe is also referred to as grassland or prairie. The steppe region of southeastern Washington is called the Palouse.

[3] A notable exception is the Juniper Dunes Wilderness Area, which preserves the northernmost Western Juniper Trees *Juniperus occidentalis* in North America. This 7100-acre area contains no surface water.

The amount of rain received by the Columbia Plateau Ecoregion each year is insufficient to support trees in the uplands/Justin Haug, WDFW

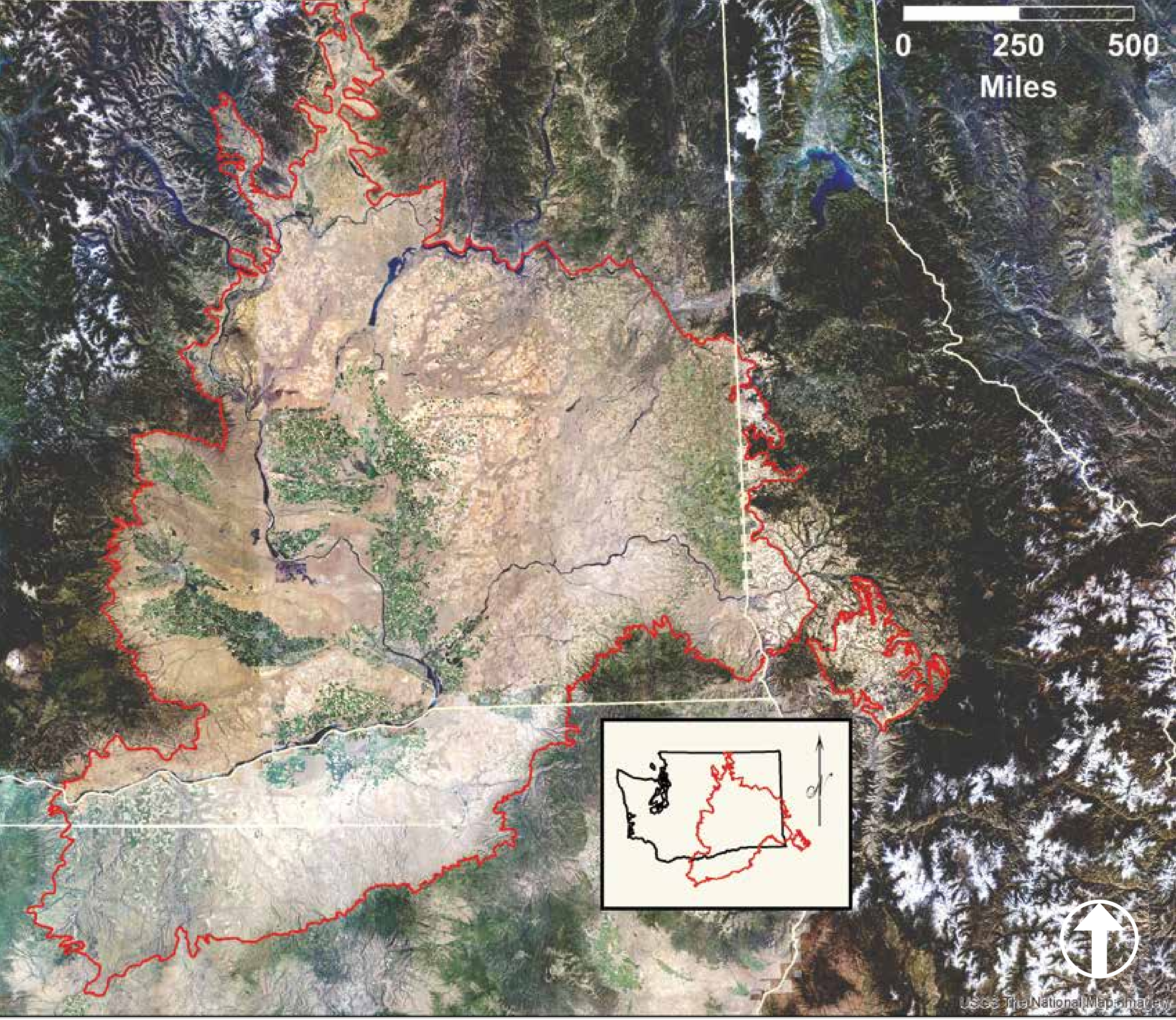

Figure 7.1. Boundary (in red) of the Columbia Plateau Ecoregion. State boundaries are white. Aerial photography for Washington State done in 2015 by the National Agriculture Imagery Program.

scientific understanding, throughout this chapter I point out differences between riparian areas in forested and unforested ecoregions.

The Columbia Plateau Ecoregion is comprised of both arid and semi-arid subregions (hereafter collectively known as drylands). Due to lack of information, I make few subregional distinctions regarding riparian areas. Most of what is known about the ecology of dryland riparian areas is based on research conducted in desert ecoregions of the southwestern United States (Patten 1998). Although these ecoregions have climate, hydrology, soils, and vegetation different from the Columbia Plateau, many of the basic principles and qualitative relationships that have emerged from research in other desert ecoregions should be valid for the Columbia Plateau.

7.1.1. Dryland Riparian Areas

Decades ago, the term "riparian" referred exclusively to areas adjacent to surface waters with moist soils that support distinctive vegetation (e.g., Thomas et al. 1979, Anderson 1987). Later definitions expanded the meaning of riparian. Sedell et al. (1989) and Naiman et al. (1992), for example, divided the "riparian ecosystem" or "riparian area" into two zones: a "riparian zone" with moist soils supporting wetland and riparian-dependent vegetation, and an upland "zone of influence" that significantly influences the exchanges of energy and matter between terrestrial and aquatic ecosystems (Figure 7.2). In forested ecoregions, the potential vegetation of both zones is commonly forest, and the ecological functions of the zones are similar. In the Columbia Plateau, vegetation within riparian ecosystems often exhibits an abrupt demarcation between the two zones. Phreatophytic[4] trees and shrubs and hydrophytic[5] herbaceous plants are confined to moist streamside areas, but the upland zone of influence may consist of sagebrush or bunchgrass communities. Consequently, the functions of the two zones may be quite different. Along some reaches, the riparian zone and zone of influence may both reside within a floodplain and exhibit similar composition and structure. For our purposes, the phrases riparian ecosystem and riparian area are synonymous.

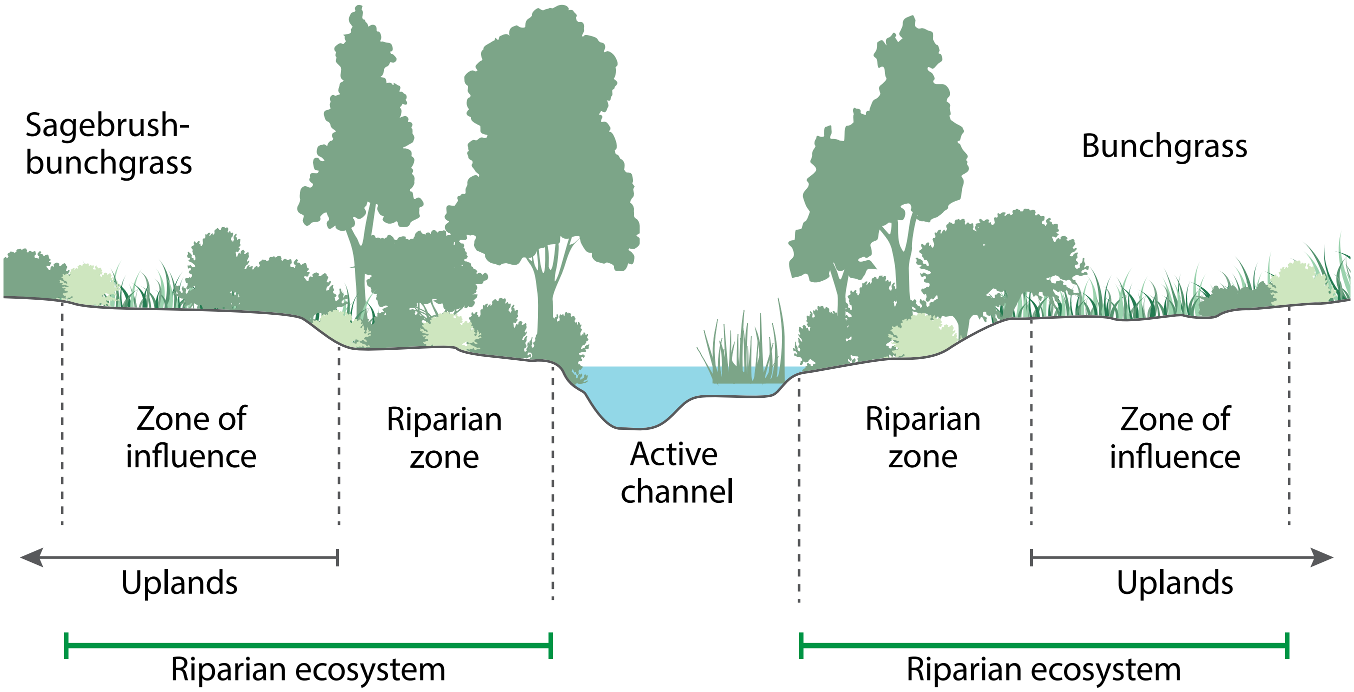

Figure 7.2. Dryland Riparian Ecosystem. The riparian ecosystem consists of two zones: riparian and zone of influence. The riparian zone extends from the edge of the active channel towards the uplands. This zone includes areas where terrestrial biota are influenced, at least periodically, by surface or subsurface waters. Beyond this is the riparian "zone of influence." This includes areas where ecological functions significantly influence the stream (diagram modified from USFS 2004).

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[4] A phreatopyhtic plant is a species that obtains water from the subsurface zone of saturation either directly or through the capillary fringe (Thomas 2014).

[5] Hydrophytic plants are those that are adapted to growing conditions associated with periodically saturated soils. They include obligate wetland plants that almost always occur in wetlands under natural conditions, facultative wetland plants that usually occur in wetlands but are occasionally found in non-wetlands, and facultative plants that equally likely to occur in wetlands or non-wetlands (Lichvar et al. 2012).

In the semi-arid Columbia Plateau Ecoregion, trees are generally confined to riparian areas/Scott Fitkin, WDFW

In dryland ecoregions, riparian ecosystems are the ultimate expression of groundwater and surface-water interactions (Webb and Leake 2006). Soil moisture and water table elevation are key variables in the survival of riparian-dependent plants. Stream discharge affects soil moisture by saturating soils and recharging the alluvial aquifer during overbank flooding, and stream stage (i.e., surface elevation) directly affects water table elevation.

In dryland ecoregions, trees and dense shrubs exist only near surface water, such as a river, stream, or lake. In the Columbia Plateau, surface streamflow is highly dependent on its source. Rivers and perennial[6] streams generally originate in the adjacent mountainous ecoregions (Omernik and Gallen 1986), and their peak flows, if unaffected by dams, generally occur during fall or winter storms and/or during spring snowmelt (Reidy Liermann et al. 2012). Within the Columbia Basin, the flow source of most streams is groundwater (Reidy Liermann et al. 2012), and consequently, during the hot, dry summer, many streams originating within the ecoregion are intermittent (Daubenmire 1942; Bryce and Omernik 1997). This spatial and temporal variability in local hydrology affects the distribution, abundance, and species of plants in dryland riparian areas.

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[6] A perennial stream is one which flows continuously. An ephemeral stream, or section of stream, is one that flows only in direct response to precipitation. It receives no water from springs and no long-continued supply from melting snow or other surface source. An intermittent stream is one that flows during protracted periods when it receives water from some surface or subsurface source, such as melting snow or a spring (Meinzer 1923).

The diversity[7] of riparian climax plant communities in the Columbia Plateau Ecoregion is greater than the diversity in surrounding forested ecoregions. In forested ecoregions, the potential natural vegetation of nearly all riparian areas along rivers and streams is forest. The species composition may vary, but the climax community's structure ultimately attains a closed-canopy of trees, with the upland zone of influence most often dominated by conifer species. In the Columbia Plateau, differences in hydrology and geomorphology manifest substantial site-level differences in composition and structure of riparian vegetation (Hough-Snee et al. 2015). A rudimentary classification system based on overstory conveys obvious differences in vertical structure: tall tree, short tree, tall shrub, shrub, grass-like, grass, and forb (Figure 7.3; Crawford 2003). The main tall tree types have an overstory consisting of Black Cottonwood *Populus trichocarpa*, White Alder *Alnus rhombifolia*, or Quaking Aspen *Populus tremuloides*. Short tree types have an overstory of Thinleaf Alder *Alnus incana*, Water Birch *Betula occidentalis*, or Black Hawthorn *Crataegus douglasii*.[8] Tree vegetation types are usually multi-layered with shrub and herbaceous layers under the tree canopy. Common shrubs are Redosier Dogwood *Cornus sericea*, Common Snowberry *Symphoicarpos albus*, Lewis' Mockorange *Philadelphus lewisii*, and Woods' Rose *Rosa woodsii*. Tall shrub vegetation types include those dominated by willow species (e.g., Yellow *Salix lutea*, Sandbar *S. exigua*) which can be up to 20 ft (6 m) tall.

The structural diversity of riparian vegetation in the Columbia Plateau causes differences in site-level functions. Large wood recruitment, for instance, cannot occur in all riparian areas, and the amount of stream shading will vary considerably due to site-to-site variation in vertical structure.

The most diverse and productive biological communities of the Columbia Plateau are riparian areas/Rex Crawford, WNHP

Nearly all woody vegetation in riparian areas of the Columbia Plateau consists of deciduous shrubs and trees/Scott Fitkin, WDFW

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[7] Measures of biological diversity take into account two factors: species richness (i.e., the number of different types) and evenness (i.e., the relative abundance of different types). While the richness of riparian climax plant communities in forested ecoregions may be comparable to that of dryland ecoregions, the relative abundance of different riparian climax plant communities is more even in dryland ecoregions. This results in a greater diversity of types.

[8] Black Hawthorn might also be considered a tall shrub. Crawford (2003) reported Black Hawthorn in riparian areas to be 6 to 20 ft (2 to 6 m) tall, and Daubenmire (1970) found Black Hawthorn up to 27 ft (8 m) tall.

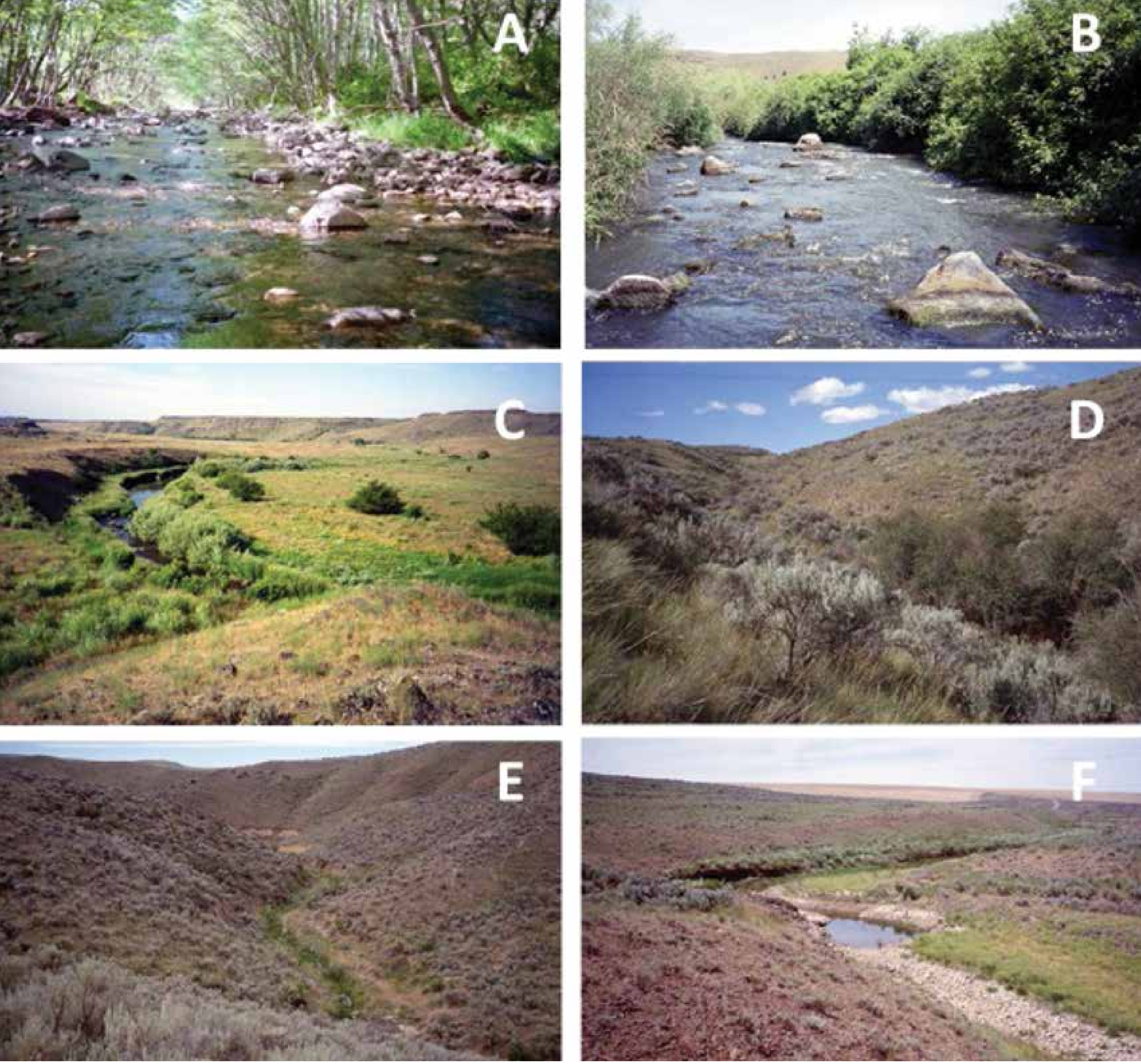

Figure 7.3. Different riparian vegetation structure types: A) tall tree, B) short tree, C) tall shrub, D) shrub on an intermittent stream, E) grass-like, F) forb/Rex Crawford, WNHP

7.2. Historical Context

Because western Washington and Oregon have a substantial amount of national park, wilderness, and roadless areas, the scientific literature contains many descriptions of unmanaged, natural riparian areas within forested ecoregions (e.g., Fonda 1974; Campbell and Franklin 1979; McKee et al. 1982; Sedell and Swanson 1984; Pabst and Spies 1999; Latterell and Naiman 2007; Fox and Bolton 2007). In the Columbia Plateau Ecoregion, however, nearly all riparian areas have been significantly impacted by human land use (grazing, intensive agriculture), resource exploitation (Beaver *Castor canadensis* trapping, timber harvest), water management (dams, diversions, reservoirs), or invasive species (Reed Canary Grass *Phalaris arundinacea*, Russian Olive *Elaeagnus angustifolia*). Daubenmire wrote in 1942, "few typical remnants of the original prairie and desert remain. These relics of the primeval vegetation likewise seem to be heading toward nearly complete extermination within a few years . . ." In short, the conditions of natural riparian areas in the Columbia Plateau Ecoregion are effectively unknown to science.

7.2.1. Beaver Trapping

The first major ecological impacts of Europeans on riparian areas in the Columbia Plateau Ecoregion were due to American Beaver (hereafter Beaver) trapping. In the early 1800s, Beaver hats were very fashionable and in high demand across Europe. The Hudson's Bay Company was determined to profit from that demand, and to obtain Beaver skins it managed a network of forts or trading posts throughout the Pacific Northwest (and throughout western North America). Several were established in and around the Columbia Plateau: Spokane House at the mouth of the Little Spokane River in 1810; Fort Okanagan at the confluence of Okanogan and Columbia rivers in 1811; a temporary post from 1812 to 1813 near confluence of Snake and Clearwater rivers; Fort Nez Perces[9] near the Walla Walla and Columbia rivers' confluence in 1818; and a temporary post at The Dalles from 1829 to 1830 (Meinig 1968). Most Beaver skins traded at these forts were obtained from the surrounding forested ecoregions.

Fort Nez Perces was situated near the center of the Columbia Plateau, and its district, which was delineated by The Dalles to the west, Priest Rapids to the north, and the Deschutes, John Day, and Grande Ronde watersheds to the south, was considered poor "fur country." Nevertheless, between 1827 and 1831 about 1,150 Beaver skins per year were traded at Fort Nez Perces (Meinig 1968). During that period, the number of Beaver skins traded per year at Fort Nez Perces dropped by 50%. Given the locations of trading posts and the high demand for Beaver skins, if Beaver could be trapped profitably in the Columbia Plateau, then they probably were. North of the Columbia Plateau, Beaver were also heavily exploited, and impacts to Beaver populations are indicated by trends in Beaver skins. The number of Beaver skins traded at the Hudson's Bay Company's Fort Colville (founded in 1825) dropped precipitously from 3600 per year in 1826 to 438 per year in 1850 (Johnson and Chance 1974).

The Hudson's Bay Company's annual Snake Country Expeditions operated out of Fort Walla Walla (formerly Fort Nez Perces). One goal of these expeditions was to create a "fur desert" which was intended to discourage America's westward expansion into lands controlled by Great Britain (Lorne 1993). George Simpson, the Hudson's Bay Company's governor at Fort Walla Walla, wrote in 1824, ". . . we have convincing proof that the country is a rich preserve of Beaver and which for

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[9] In 1818 the North West Company opened a fur-trading post near the mouth of the Walla Walla River at the current location of Wallula, Washington. Originally called Fort Nez Perces, it was renamed Fort Walla Walla when the North West Company and the Hudson's Bay Company merged in 1821 (Denfeld 2011).

political reasons we should endeavor to destroy as fast as possible" (Ott 2003). In 1825 the trapper Peter Ogden and his party collected 3,577 Beaver skins in the Crooked River watershed, a semi-arid portion of the neighboring Blue Mountains Ecoregion, and on two subsequent expeditions trapped more Beaver in this area (Buckley 1993). During the 1825 expedition, Odgen noted that the Crooked River's tributaries were well lined with willows and aspen. By 1910, if not sooner, Beaver had been eliminated from some tributaries of the Crooked River, and the willow and aspen had vanished (Buckley 1993). The eradication of Beaver from dryland riparian areas is thought to have caused a cascade of adverse effects upon riparian ecosystems that I discuss in Section 7.3.

7.2.2. Open Range Grazing by Livestock

Open-range grazing by livestock was the next major ecological impact of Europeans in the Columbia Plateau Ecoregion. Around 1730, Native Americans on the Columbia Plateau acquired horses descended from those brought by Spanish explorers (Carlson 1940). In 1805, Lewis and Clark saw about 700 horses in one Indian village and thousands more in the nearby hills (Galbraith and Anderson 1971). Daubenmire (1970) believed that while most villages were on rivers, the impacts to riparian vegetation by horses were probably not extensive. A few cattle were brought to the Okanogan Valley in 1825 and to the Walla Walla Valley in 1834 (Carlson 1940). By 1855, roughly 200,000 cattle inhabited the Columbia Plateau of Washington (Daubenmire 1970). Sheep herding developed in the 1880s (Daubenmire 1970), and great numbers of sheep were brought to eastern Washington in 1892 (Galbraith and Anderson 1971). By the late 1800s, damage to rangeland from overgrazing was evident. Based on his 1893 expedition from Spokane to Steven's Pass, the botanist John Leiberg wrote, "We will never know the complete flora of these regions. Sheep and cattle are rapidly destroying the native plants and by the time private explorations reach these regions the flora will have been totally exterminated by such agencies" (as cited in Mack 1988; Weddell 2001).

Open-range management of livestock invariably results in excessive utilization of riparian areas (Ohmart 1996), and the destructive impacts of unmanaged grazing in riparian areas are well-documented (Kauffman and Krueger 1984; Ohmart 1996, Belsky et al. 1999). Riparian areas in arid and semi-arid lands of the western United States cover only 1 to 2% of land area, but produce 20% of available forage and nearly 80% of forage actually consumed by cattle (Fisher 1995). Therefore, it is reasonable to assume that the damage observed by Leiberg was greatest in riparian areas.

Restoration projects can indicate the density of woody plants that existed historically in riparian areas prior to intensive livestock grazing. For example, two years after the cessation of grazing within riparian areas in northeastern Oregon, the mean crown volume of willows and Thinleaf Alder tripled in size and that of Black Cottonwood increased nine-fold (Case and Kauffman 1997). Furthermore, shrub density increased by 50%. One possible reason for the severe impacts of grazing in the Columbia Plateau is that, unlike the Great Plains, Bison (*Bison bison*) have been rare or absent from the Columbia Plateau for close to 10,000 years (Jones 2000). According to Daubenmire (1970) and also Mack and Thompson (1982), ungulates played no significant role in the evolution of plant species in steppe and shrub-steppe communities of the Columbia Plateau, and consequently present-day vegetation cannot endure intensive grazing by livestock.

The impacts of both Beaver trapping and open-range livestock grazing may have acted synergistically to degrade riparian areas. Baker et al. (2005) suggest that Beaver and willow (*Salix* spp.) are mutualists because willow is an important food source for Beaver, and Beaver create environmental conditions suitable as willow habitat. Baker et al. (2005) believed this relationship can persist indefinitely within a given area. They also speculated that heavy browsing by native ungulates or livestock disrupt the Beaver-willow mutualism, thus leading to degradation of the riparian ecosystem. Ohmart (1996), based on personal observation, suggested that overgrazing by livestock causes an imbalance in the Beaver-vegetation relationship that leads to collapse of riparian forests. A similar relationship has also been observed in Yellowstone National Park (Beschta and Ripple 2016) where decades of intensive grazing by Elk *Cervus canadensis* during the absence of Wolves *Canis lupus* resulted in riparian plant communities that could no longer support Beaver. The reintroduction of Wolves has led to a recovery of Beaver in Yellowstone (Beschta and Ripple 2016).

Today in the Columbia Plateau Ecoregion, the most severe impacts from grazing are caused by wild or feral horses (hereafter collectively known as wild horses). Until the mid-1900s, wild horses inhabited open rangelands in various parts of the Columbia Plateau. Today wild horses are confined to Native American reservations where the number of wild horses has fluctuated greatly over the past 140 years. The reservation of the Yakama Nation, for instance, supported roughly 16,000 horses in 1878, but a round up in 1957 left 300 horses on the reservation (Adams 2004). By 2010, the number of horses had rebounded to 12,000 (YN 2010), but the current carrying capacity of the Yakama Nation's reservation is only about 1,000 horses (Mapes 2010). Consequently, wild horses have destroyed rangelands, cultural resources, and wildlife habitats, including riparian areas (NTHC, no date). Wild horse populations also exceeded the carrying capacities of the Colville and Umatilla reservations (AP 2012; Tribal Tribune 2013). Effective management of wild horses on Native American reservations has been hampered by continual controversy regarding the most practical and ethical means of reducing their population sizes.

7.2.3. Wood Harvest and Wheat

The Donation Land Claim Act of 1850 and Homestead Act in 1862 were major impetuses for settlement of Oregon and Washington states. Settlers needed wood for fuel and construction. On the Columbia Plateau the closest source of wood, and in some places the only practical source, was riparian areas. Historians have little information on wood harvest in riparian areas by early settlers on the Columbia Plateau, however, the practices of pioneers and settlers on the Great Plains, as described by West and Ruark (2004), are likely to be similar to their practices on the Columbia Plateau. West and Ruark (2004) state that pioneers on their way west stripped virtually all trees from the Platte River valley, that riparian areas on the Great Plains were an important source of winter fuel for settlers, and that by 1900 most of the trees in riparian areas of the Great Plains had been harvested. Daubenmire (1970) speculated that Black Cottonwood, the largest tree species in the Columbia Plateau, was much more abundant before the arrival of settlers.

Impacts of Beaver trapping and open-range livestock grazing may have acted synergistically to degrade riparian areas. Baker et al. (2005) suggest that Beaver and willow are mutualists because willow is an important food source for Beaver, and Beaver create environmental conditions suitable as willow habitat.

River commerce on the Columbia and Snake rivers may have also impacted riparian areas. By 1859, steamboats were travelling upriver as far as Wallula, Washington and could reach Lewiston, Idaho at high water (Meinig 1968, cited in Evans 1989). HIstorians do not know how much wood was burned to power steamboats in the Columbia Basin, but estimates from other western watersheds suggest severe impacts to riparian forests along the Columbia and Snake. On the upper Missouri River in Montana, for instance, riparian forests along its banks were denuded of usable trees by the turn of the century (Evans 1989).

The soils and climate of the Palouse Hills subregion are suitable for dryland wheat agriculture. Consequently, the next major impact to riparian areas was converting native vegetation to wheat. The first wheat was planted in 1877, by 1895 most of the tillable land in the subregion had been plowed (Kaiser 1961), and during the 1970s industrial agriculture expedited the destruction of the last refugia for native plant communities (Black et al. 1998). Agriculture in the Palouse Hills had four major impacts on riparian areas, which were all poorly documented: 1) fields were plowed to the very edge of streambanks, thereby destroying riparian vegetation, 2) many ephemeral channels were plowed through and filled (Kaiser 1961; Bryce and Omernik 1997), 3) perennial and intermittent stream channels became deeply incised (Rockie 1939), and 4) some channels were filled with sediment from soil erosion (Rockie 1939; Bryce and Omernik 1997). Rockie (1939) reported a Nez Percé Indian telling him that some dry channels in the Palouse Hills were once perennial streams containing trout. According to Bryce and Omernik (1997), the original channel substrate, riparian vegetation, and flow regime (i.e., perennial or intermittent) are impossible to know in areas of intensive agriculture.

The least arid subregions of the Columbia Plateau support productive grasslands/Alan Bauer

7.2.4. Large-scale Irrigation Projects

Across the Columbia Plateau, the most obvious human impacts to native habitat types, including riparian areas, are those associated with irrigation projects. The U.S. Bureau of Reclamation manages two major projects on the Columbia Plateau, the Yakima and Columbia Basin, which service about 1,135,000 acres (459,318 ha) of farmland (USBR 2017a; 2017b). Another 37,000 acres (14,973 ha) are irrigated with water impounded by dams on the Snake River (USACE 2002). All 1.17 million acres (473,482 ha) of irrigated farmland were once shrub-steppe or steppe vegetation. However, these projects did not record how many acres of riparian area have been lost because ephemeral or intermittent stream channels were filled for agriculture or converted to irrigation ditches.

Construction of the Yakima Project began in 1906, and the last of 11 dams on the Yakima River or its tributaries was completed in 1939 (USBR 2017a). The Yakima Project's biggest impacts to riparian areas are along the Naches and Yakima rivers. Storage reservoirs and water diversions located upstream have truncated the rivers' natural hydrographs. This has substantially reduced river-floodplain interactions, such as inundation and lateral channel migration, which in turn have led to channel simplification and degraded aquatic and riparian habitats (Snyder and Stanford 2001; YSPB 2004).

Construction of the Columbia Basin Project began in 1933 with Grand Coulee Dam, and the first irrigation water was delivered in 1952 (USBR 2017b). The Columbia Basin Project includes a network of canals, drains, and wasteways over 5,800 miles (9,334 km) long. The reservoir behind Grand Coulee Dam, Lake Roosevelt, inundated roughly 84 miles (135 km) of riparian habitat within the Columbia Plateau Ecoregion along the Columbia, Spokane, and Sanpoil rivers.[10] Roughly 460 miles (740 km) of the Columbia River lie within the Columbia Plateau Ecoregion, but only 51 miles (82 km) of riparian habitat, the Hanford Reach, have not been inundated by reservoirs. When the dams were built, there was apparently some hope that vegetation would re-establish along new reservoir margins, but water level fluctuations appear to be too extreme for most plants to tolerate (Evans 1989). The other major impacts to riparian areas from the Columbia Basin Project are dramatic changes in hydrology. The groundwater table around lower Crab Creek, for instance, has risen between 50 and 150 ft (15 and 46 m) since 1952. Furthermore, the formerly intermittent lower Crab Creek is now perennial with four to five times the flow that occurred prior to irrigation (KWA 2004), and, as a consequence, willow species, which are adapted to the natural hydrologic regime, no longer regenerate along lower Crab Creek (Ortolano and Cushing 2000). In contrast, due to groundwater withdrawals, the groundwater table around upper Crab Creek (the Odessa subarea) has fallen 150 ft (46 m) (KWA 2004). I found no records on how the water table changes may have affected stream flow in upper Crab Creek.

Decades of hydrological alteration by dams and water diversions have altered vegetation composition and structure of riparian areas. Black Cottonwood and willow are successional pioneers that require barren sediments of disturbed alluvial floodplains for seedling germination (Patten 1998). Such conditions are produced by floods, but dams reduce peak flows and the erosive forces that scour sediments. Furthermore, the survival and growth of Black Cottonwood seedlings is highly dependent on water table elevation and its seasonal rate of change (Braatne et al. 1996). If the water table declines more rapidly than seedling root growth, then seedlings die. Braatne et al. (2007), for example, found that dams on the Yakima River created

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[10] Estimated with Google Earth. Distance estimated along Spokane River comports with that reported by Ortolano and Cushing (2000).

hydrological conditions unsuitable for Black Cottonwood seedlings. They studied river reaches near Cle Elum, Union Gap, and Wapato and found: 1) no seedlings in any reaches, 2) Black Cottonwood stands less than 25 years old were extremely limited, 3) riparian areas were dominated by older age classes, and 4) altered hydrology promoted invasion of the floodplain by exotic tree species such as Silver Maple *Acer saccharinum*, Norway Maple *Acer platanoides*, and Chinese Elm *Ulmus parvifolia*. Hence, one could confidently infer that historical conditions have been altered by regulated flows. If artificial flow regimes are not changed, then as older individuals senesce and die, cottonwoods may be eliminated from some sections of the Yakima River. This would have adverse effects on multiple riparian area functions: streambank stability, shading, large wood recruitment, and detrital nutrients. Exotic tree species currently found along the Yakima River cannot provide the same ecological functions as Black Cottonwood because they cannot establish themselves in the barren, saturated sediments of recently disturbed floodplains.

Dams have also altered the flow of wood through watersheds. Historically, wood was transported from forested headwaters to dryland reaches of rivers and streams. This process is known to have occurred in other semi-arid and arid regions of the United States. Minckley and Rinne (1985) present historical evidence for the movement of large wood in desert rivers of the American southwest: wood originated in forested headwaters, moved sporadically through desert riparian areas during flood events, and was ultimately deposited at the mouth of the Colorado River in Mexico. Minckley and Rinne (1985) identified interception of large wood by dams as a major cause of large wood reduction in semi-arid and arid river basins. At artificial reservoirs in eastern Washington, such as Keechelus, Kachess, Cle Elum, Easton, Bumping, Clear, and Rimrock lakes, large wood that could potentially interfere with dam operations is removed and burned (W. Meyer, Washington Department of Fish and Wildlife, personal communication; B. Renfrow, Washington Department of Fish and Wildlife, personal communication). Large wood in dryland sections of the Wenatchee and Okanogan rivers may be effected in this way as well. How wood management at dams and smaller water diversions affects in-stream large wood in eastern Washington has not been investigated.

> *"We will never know the complete flora of these regions. Sheep and cattle are rapidly destroying the native plants and by the time private explorations reach these regions the flora will have been totally exterminated by such agencies."*
>
> - Botanist John Leiberg, 1893

7.2.5. Invasive Non-native Plants

An ongoing historical anthropogenic impact is the introduction of exotic invasive plant species. After their botanical survey of the Columbia Plateau in 1893, Sandberg and Leiberg were not troubled by the meager abundance of exotic species at that time (Mack 1988), however, by 1929 about 200 exotic species were known to exist on the plateau (Mack 1988). Riparian areas in dryland landscapes are more vulnerable to invasion by exotic plant species than adjacent upland areas (Loope et al. 1988; Hood and Naiman 2000), and invasive plants in riparian areas can impact a variety of ecological processes: hydrology, sediment dynamics, nutrient dynamics, and species competition. Two invasive species of particular concern are Reed Canary Grass and Russian Olive.[11]

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[11] Other invasive species of concern in the Columbia Plateau Ecoregion are Crack Willow *Salix fragilis*, Yellow Flag Iris *Iris pseudacorus*, and Purple Loosestrife *Lythrum salicaria*.

Reed Canary Grass occurs as both exotic and indigenous strains (Tu 2006). The European cultivar has been present in the Pacific Northwest since the 1880s (Stannard and Crowder 2001), and, as of 2016, it is found in all Washington counties except Douglas (WSDA 2016). Reed Canary Grass forms dense stands that can exclude all other plant species, and such stands exhibit substantially higher transpiration per unit ground surface area than other species (Gebauer et al. 2015). This could alter water availability in riparian areas. Reed Canary Grass also has higher roughness values than native sedge species (Martinez and McDowell 2016). This could lead to greater sediment deposition and channel narrowing.

As of 2016, Russian Olive is found in most counties east of the Cascade crest and is most common in Grant County (WSDA 2016). In the late 1800s, it was introduced into the Pacific Northwest as an ornamental plant (Giblin 2006), but the species did not become prominent outside cultivated areas until two to five decades later, depending on location. Unlike many tree and shrub species in dryland riparian areas, Russian Olive does not require disturbed soils for germination. Hence, it can grow on sites where woody vegetation would not naturally occur (Katz and Shafroth 2003). It is also shade tolerant, and consequently, Russian Olive can establish beneath the canopy of native riparian trees or shrubs and compete with them for resources. Russian Olive exhibits about two times higher transpiration per unit basal area than cottonwood trees (Hultine and Bush 2011). This could alter water availability in riparian areas. The species is a nitrogen fixer that adds nitrogen to the soil. Tuttle et al. (2016) found that stands of Russian Olive had higher exotic plant ground cover than stands of native tree species, and they believed that the difference was due to higher available soil nitrogen that benefited fast growing exotics plants. Finally, according to Lesica and Miles (1999), Beaver foraging behavior indicates that Russian Olive is much less palatable than native cottonwood. They found that 77% of cottonwood trees along a river in Montana were damaged by Beaver but only 22% of Russian Olive trees were damaged. If Russian Olive replaces native cottonwood, then Beaver population density could be adversely affected.

7.2.6. Value of a Historical Perspective

Current scientific understanding of riparian ecosystems in the Columbia Plateau Ecoregion is mostly based on present-day conditions, which are unlikely to accurately represent historical composition and structure. History indicates that past human impacts in the Columbia Plateau may have erased fully functioning riparian ecosystems from many watersheds. For example, records of the U.S. Army cavalry at Fort Simcoe on the Yakama Indian Reservation from the 1850s describe small valleys with broad floodplains dissected by multi-threaded stream channels that are absent today. These valleys—such as the Wenas, Ahtanum, and Cowiche—historically contained a network of streams, wetlands, and Beaver ponds that supported extensive groves of willow, Redosier Dogwood, Black Cottonwood, and aspen. Cattle were said to be lost for weeks in these tangles (P. Harvester, Washington Department of Fish and Wildlife, personal communication). These same streams are now reduced to a single, incised channel with minimal riparian vegetation. The loss of riparian habitats indicated by the Fort Simcoe records is very likely to have occurred in other parts of the Columbia Plateau.

Rivers and streams of the Columbia Plateau Ecoregion have been degraded by a variety of human activities. This has adversely affected anadromous salmon—their populations are threatened with extinction and fisheries produce a small fraction of historical harvest. In response, salmon habitat restoration projects are occurring throughout the Columbia Plateau. Habitat restoration requires ecological reference information to determine site-level restoration potential, establish desired conditions, and formulate measures of success

(White and Walker 1997). Ecological reference information is generally based on historical conditions, natural areas, or minimally disturbed sites (White and Walker 1997; Stoddard et al. 2006; Higgs et al. 2014). In the Columbia Plateau Ecoregion, which lacks national parks, wilderness, or roadless areas containing natural reference sites, historical information may be the only basis for reference conditions (Wohl 2005).

Because the Columbia Plateau lacks undisturbed watersheds, even small ones, ecologists have a poor understanding of the composition, structure, and functions of natural riparian ecosystems. Historical reconstruction using journals of early explorers and naturalists, General Land Office survey notes, and historical drawings or photos (Figure 7.4) are ways to develop qualitative descriptions of riparian plant

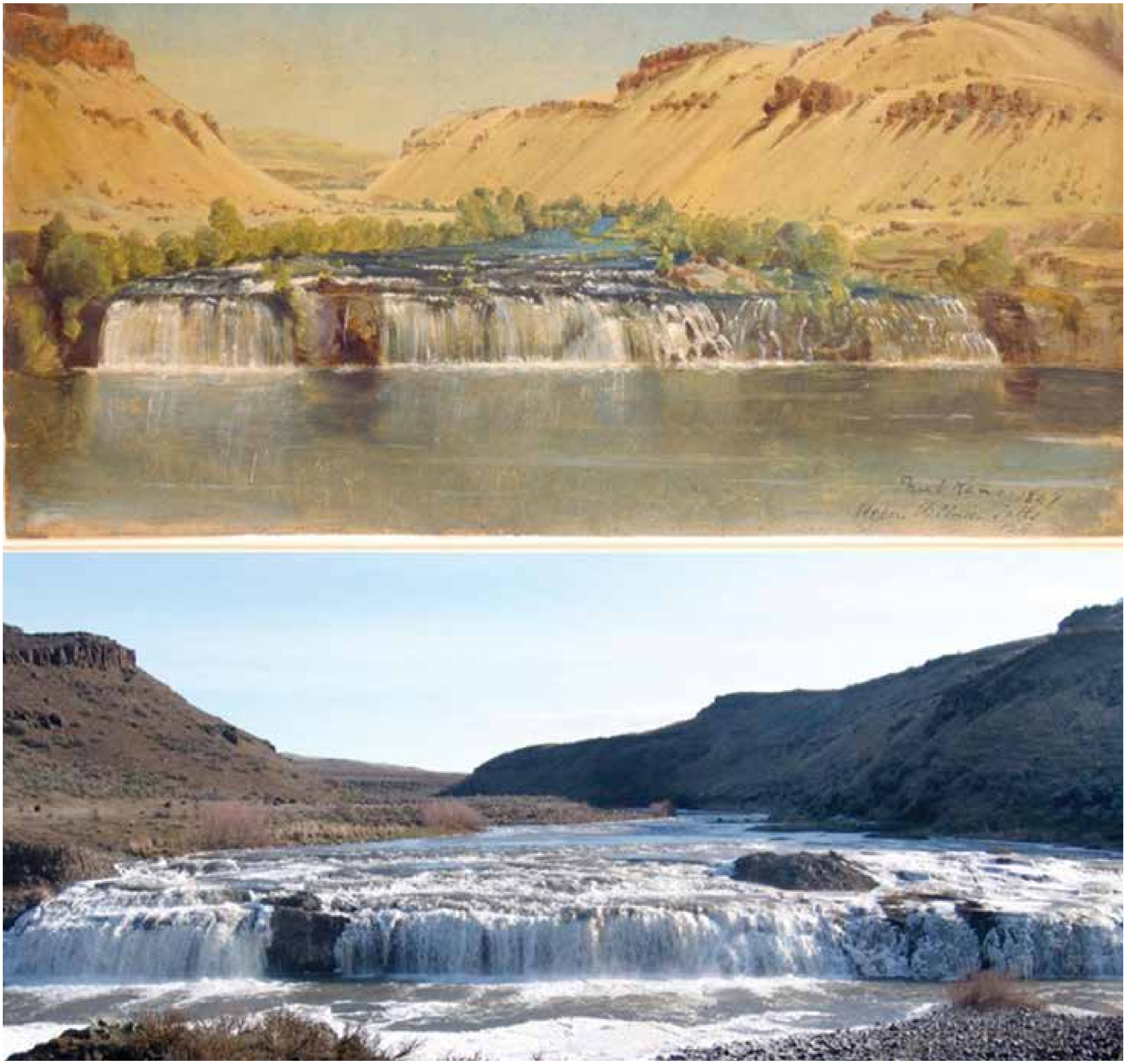

Figure 7.4. Historical drawings and paintings can provide invaluable information for riparian area restoration. A 1847 painting by Paul Kane of Upper Palouse Falls (top) shows extensive, uninterrupted small tree or tall shrub vegetation along the river banks. A recent photo by Jack Nisbet (bottom) although taken in winter, shows large gaps in small tree or tall shrub vegetation along the banks.

communities (McAllister 2008). Much work needs to be done on deducing the likely historical conditions of riparian areas in the Columbia Plateau. Studies of historical conditions in the Columbia Plateau, however, must be cognizant of one important caveat. If hydrology of a riparian area has been significantly altered by human activity, then the vegetation that site supported historically may have no relationship to the vegetation the site is capable of supporting now. Irrigation systems have altered hydrology in riparian areas throughout the Columbia Plateau, making some drier and others wetter. At such places, information on historical vegetation will often be of little value in establishing desired future conditions for restoration.

7.3. Beaver

Beaver may play an essential role in the restoration and future conservation of dryland riparian areas (Dremmer and Beschta 2008; Pollock et al. 2014). The ecosystem benefits of Beaver in dryland riparian areas have been known for over 80 years. Scheffer (1938) reported that in 1936 the Soil Conservation Service transplanted 10 Beaver to the Ahtanum Creek watershed for the purposes of increasing water storage. The Beaver constructed several large dams, one of which was 90 ft (27 m) long, 10 ft (3 m) high, and stored 5 acre-feet (6,167 m^3) of water. Scheffer (1938) implied that the dams reduced downstream flood damage and increased summer stream flows for irrigation. Finley (1937) reported the opposite situation for Silver Creek in eastern Oregon. Two trappers removed 600 Beaver in one winter. As a result, the Beaver ponds “disappeared,” the water table lowered, grassy meadows “died out” and streams containing trout went dry.

The observations of Scheffer (1938) and Finley (1937) are anecdotes. More rigorous scientific research into the connections between Beaver and riparian ecosystems have found that Beaver have numerous positive effects.

Beaver viewed with night vision camera/Ned Pittman, WDFW

Beaver dams store organic nutrients and large quantities of sediment, reduce channel incision, remove excess nutrients from water, increase surface water storage and base flows, reduce peak flows, increase groundwater recharge, raise water table elevations, expand the area of riparian vegetation, and increase the salmonid habitat capacity of small streams (Naiman et al. 1988; Pollock et al. 2003; Gibson and Olden 2014). For dryland riparian areas where the extent of riparian vegetation depends on shallow groundwater, the hydrologic and geomorphic effects of Beaver dams may be necessary for the mere existence of a riparian zone at many sites. In dryland riparian areas, most ecosystem functions occur within the riparian zone.

The dramatic effects of Beaver on hydrology was shown by Westbrook et al. (2006), a multi-year study of two Beavers dams on a fourth order stream running through a dry meadow in the Rocky Mountains of Colorado. One dam diverted over half and the other diverted 70% of the stream onto the floodplain and terrace. Both dams increased substantially the area flooded by annual peak flows. One dam caused the equivalent of a 20-year flood over 30 acres (12 ha) of the valley floor during a peak flow event with a 1.6-year recurrence interval, and the other dam caused the equivalent of a 200-year flood over 21 acres (8 ha) of the valley floor during a peak flow event with a 1-year recurrence interval. One of the dams studied by Westbrook et al. (2006) was breached by high flows. Westbrook et al. (2011) subsequently found that overbank flooding caused by that dam had deposited nutrient-rich sediments in some areas and scoured soils in other areas downstream of the dam site. These bare sediments were colonized by willow and aspen seedlings, which may eventually establish riparian plant communities.

In sagebrush steppe of Wyoming, Cooke and Zach (2008) found that the number of Beaver dams was positively associated with riparian zone width, riparian shrub height and percent cover by emergent wetland vegetation. These responses can at least partly be explained by changes in hydrology caused by Beaver dams.

Eradication of Beaver has been implicated in stream channel incision and desertification of riparian zones that has occurred throughout arid and semi-arid ecoregions of western North American (Parker et al. 1985; Pollock et al. 2003), including the Columbia Plateau (Pollock et al. 2007). According to Pollock et al. (2007), the exact mechanism that caused widespread incision of streambeds remains uncertain, however, incision almost invariably coincided with widespread trapping of Beaver and the onset of intensive livestock grazing. Channel incision, whatever the ultimate cause, occurs through an imbalance between sediment deposition and sediment mobilization. Assuming no changes in upstream sediment sources, incision is usually attributed to an increase in erosive forces or a decrease in channel resistance to erosive forces. The former can be caused by an increase in runoff, both can be caused by a decrease in channel roughness, and the latter can also be caused by loss of streamside vegetation. Therefore, channel incision can be prevented by controlling runoff caused by upstream land uses, maintaining roughness elements such as large wood, and retaining riparian vegetation. For many dryland riparian areas, Beaver are essential for the latter two. Beaver dams act as massive roughness elements that reduce flow velocities, thereby reducing hydraulic forces that cause erosion. Reducing flow velocities also results in sediment deposition, which raises the streambed elevation and creates microsites suitable for germination of riparian plants.

Several studies have suggested that reintroduction of Beaver can reverse channel incision (Pollock et al. 2007; Beechie et al. 2008), primarily by restoring sediment retention processes in the channel (Pollock et al. 2014). Sediment aggradation caused by Beaver dams fills the incision and raises the streambed elevation. As the bed elevation increases, hydrologic connectivity between the channel and former floodplain is improved both by raising the water table elevation and by reducing bank height, which allows overbank flooding to recharge the floodplain's alluvial aquifer. Increased soil moisture allows riparian plant species vegetation to re-establish. Development of a well-vegetated riparian zone provides Beaver with more food and building materials, enabling construction of more dams (Pollock et al. 2014).

The aforementioned impacts of Beaver on aquatic and riparian ecosystems are immensely beneficial to fish; however, Beaver may also have detrimental effects on fish. Sediment deposition in Beaver ponds may harm spawning beds, and Beaver dams can block upstream and downstream fish movements (Pollock et al. 2003). Water temperature in Beaver ponds can be warmer than nearby flowing waters, especially where the stream is largely under a closed canopy forest (Robison et al. 1999), but in areas with open or no canopy (such as dryland riparian areas), water temperatures tend to be similar between Beaver ponds and unimpounded reaches (Talabere 2002). Beaver ponds have also been found to provide productive habitats for non-native fish (Gibson et al. 2015) that may compete with or prey upon native fish.

7.4. Ecosystem Functions

FEMAT (1993) identified five key functions of riparian ecosystems that are important for fish habitats: streambank integrity affected by plant roots, litter fall that provides detrital nutrients, shading to limit stream temperatures, large wood recruitment, and pollution removal. Most of what is known about these key riparian functions was discovered through research conducted in forested ecoregions, in particular, the mesic forest ecoregions of the Pacific Northwest (e.g., Salo and Cundy 1987; Naiman and Bilby 1998; Richardson et al. 2005). Much of what is known about riparian areas in forested ecoregions is transferable to riparian areas in the Columbia Plateau, and therefore, where appropriate I repeat some of the knowledge and principles elucidated in chapters 1 through 6. However, as will become apparent, large gaps in knowledge remain.

In addition to FEMAT's key ecological functions, a sixth key function occurs in dryland riparian areas—alluvial water storage. Alluvial water storage occurs in riparian areas of all ecoregions, but it may be especially important in dryland ecoregions.

7.4.1. Streambank Stability

Streambank stability refers to a bank's resistance to change and its resilience after change (Bohn 1986). Streambank stability is the opposite of streambank erosion. Streambank erosion is commonly misinterpreted as a sign of adverse human impacts to riparian ecosystems, and it can be. However, a channel in dynamic equilibrium simultaneously exhibits both stable and unstable banks (Wohl et al. 2015). Streambank erosion is integral to a watershed's fluvial disturbance regime and necessary for long-term ecological sustainability (Florsheim et al. 2008) because the opposing processes of bank stability and bank erosion maintain the diversity of aquatic and riparian habitats within a watershed. Bank stability is also associated with overhanging/undercut banks which shade the water and provide cover for fish (Bohn 1986; Ohmart 1996).

The stability of streambanks is influenced by soil characteristics, groundwater, and vegetation. For the purposes of riparian area management, I focus on vegetation. Rooted vegetation is essential for streambank stability (Hupp and Osterkamp 1996; Gurnell 2014). Plants are said to provide "root strength" or "root reinforcement" along streambanks (FEMAT 1993; Pollen-Bankhead and Simon 2010). Soil is generally strong in compression, but weak in tension. Woody and herbaceous plant roots are strong in tension, but weak in compression. Consequently, the root-permeated soil of streambanks behaves as a composite material with enhanced strength (Simon and Collision 2002). Dense root networks also physically restrain or bind soil particles. Exposed roots on the bank surface increase channel roughness which dampens stream flow velocities, thereby reducing fluvial erosion (Griffin et al. 2005). Reduction of stream velocities by roots may also cause sediment deposition, which further stabilizes streambanks.

The stability provided by roots is species dependent (Polvi et al. 2014). Simon et al. (2006), for instance, found that Lemmon's Willow *Salix lemmonii* provided 10 times more root reinforcement of streambanks than Lodgepole Pine *Pinus contorta.* The difference was due to the greater root density and larger root area of Lemmon's Willow. On the southeastern slope of the Cascades Range in Washington, Liquori and Jackson (2001) found that different riparian plant communities manifested differences in bank erosion and channel form. They identified two types of riparian plant communities: a scrub-shrub community that exists within an open-canopy Ponderosa Pine *Pinus ponderosa* forest, and a dense, closed-canopy fir forest. The

Ponderosa Pine forest is maintained by regular wildfires while the fir forest was created by fire suppression. The scrub-shrub community was associated with stream channels that had less bank erosion, lower width-to-depth ratios, more pools, and more undercut banks than channels within the fir forest community. The channel morphology of scrub-shrub riparian communities also resulted in slightly lower summer stream temperatures. In the semi-arid Sierra Nevada Mountains, Micheli and Kirchner (2002) found that riparian areas comprised of sedges and/or rushes had banks five times stronger than riparian areas comprised of sagebrush and grasses, and that bank strength was correlated with the root mass to soil mass ratio. Sedges and/or rushes enabled the formation of undercut banks, and increased the width of an undercut bank by a factor of 10.

The composition and structure of riparian vegetation affects channel morphology which, in turn, affects riparian vegetation (Corenblit et al. 2007; Osterkamp and Hupp 2010). Toledo and Kauffman (2001) observed the effects of one such interactive feedback loop in semi-arid northeastern Oregon. Upland plants were found along incised channels and wetland-obligate plants (mostly sedges) were found along unincised channels, which had two times greater root biomass than incised channels. Along unincised channels, approximately 40% of root biomass was deeper than 8 in (20 cm) in the soil, but along incised channels less than 25% of root biomass was below 8 in (20 cm). Denser roots at deeper soil depths result in a larger volume of reinforced soil. According to Toledo and Kauffman (2001), the severely degraded conditions at their incised sites were caused by a positive feedback loop in which an initial channel incision led to a lower water table. The resulting reduction in soil moisture altered riparian vegetation which shifted to an upland plant community with reduced root mass. Bank erosion subsequently increased which led to channel widening and further incision.

Compared to forested ecoregions, there is less large woody debris, and woody debris play a much smaller role in the fluvial processes of dryland river and streams.

The studies reviewed thus far have reported on the bank strength provided by hydric or wetland-obligate shrubs and herbaceous plants (sedges and rushes) in dryland riparian areas. Trees are also important for bank stability. Rood et al. (2015) studied the amount of bank erosion after major floods along Elk River in British Columbia. They found that floodplain locations occupied by grassland or small shrubs exhibited substantial erosion (>75 m of bank erosion) after major floods but locations occupied by forest, including Black Cottonwood stands, exhibited much less change (<15 m of bank erosion). Rood et al. (2015) concluded that "only big plants can resist big flows," and that other studies which demonstrated that grasses or shrubs are more erosion resistant had observed only low energy flows in smaller or shallower gradient streams.

Streambanks along incised channels of arid or semi-arid regions are particularly sensitive to changes in soil moisture. Much of the bank may be above the water table, and hence soils are unsaturated. Matric[12] suction above the water table, enhanced by transpiration of riparian vegetation, has the effect of increasing the apparent soil cohesion, and decreases in soil strength due to a loss of suction are a leading cause of bank failures in incised channels (Simon and Collison 2002).

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[12] Matric suction is a negative pore pressure that induces water to flow in unsaturated soil. It results from the combined effects of adsorption and capillarity of the soil matrix. Water flows from a soil with low matric suction (a wet soil) to soil with a high matric suction (a dry soil).

7.4.2. Wood

The main role of wood in aquatic ecosystems is roughness element (Bisson et al. 1987). Roughness elements are obstacles in a channel that deflect flow and change its velocity. The size, shape, and strength of large wood make it very effective at redirecting hydraulic forces and the flow of materials, such as sediment and fine organic matter. In-stream large wood increases hydraulic complexity, i.e., creates a wider range of flow velocities, which causes pool formation, streambed scour, sediment deposition, and channel migration. The net result is a diversity of aquatic habitat types.

I could find no studies conducted in the Columbia Plateau describing the density of trees or their spatial distribution in riparian areas. However, riparian areas in the Columbia Plateau certainly contain less large wood than forested ecoregions, even historically, for three reasons. First, trees are smaller in the Columbia Plateau than in forested ecoregions of Washington. Canopy dominant trees in old-growth riparian ecosystems of mesic forested ecoregions can be well over 48 in (122 cm) diameter at breast height (dbh) and over 150 ft (46 m) tall, but the maximum size of Black Cottonwood in the Columbia Plateau is much smaller (see below). Second, trees are fewer in the Columbia Plateau because in the Columbia Plateau trees inhabit the riparian zone but often do not inhabit the zone of influence. In forested ecoregions, trees inhabit both zones. Third, along some streams, particularly in the most arid subregions of the Columbia Plateau, even the riparian zone is unsuitable for tree species. At such sites, the riparian zone consists of tall shrub, shrub, grass-like, grass, and forb vegetation types. Therefore, compared to forested ecoregions, there is less large woody debris, and woody debris plays a much smaller role in the fluvial processes of dryland river and streams.

Knowledge regarding wood in aquatic and riparian ecosystems of Columbia Plateau is extremely limited at present, nevertheless, the historical abundance of woody plants in riparian areas is thought to have been much greater than it is today (Wissmar et al. 1994; Kauffman et al. 1997; Wissmar 2004). Riparian areas in the Columbia Plateau are inhabited by a variety of woody plants—Black Cottonwood, Quaking Aspen, White Alder, Thinleaf Alder, Water Birch, Black Hawthorn, and Yellow Willow (Crawford 2003). Conifer trees, including Ponderosa Pine and Douglas-fir *Pseudotsuga menziesii,* are widely scattered in eastern Washington riparian areas and were likely more common historically than at present. They are currently restricted to canyons or valleys with steep rocky walls along mid- to high-gradient streams where they are inaccessible to harvest and where microclimates are conducive to supporting trees (Evans 1989). The main causes of reduced wood are explained in Section 7.2.

The size and quantity of wood recruited to a stream depends on the size and density of woody plants, their distances from the channel, and the recruitment processes that deliver wood to the channel. Only a few tree species in riparian areas of the Columbia Plateau qualify as large wood, which is commonly defined as greater than 4 in (10 cm) in diameter and greater than 6 ft (2 m) in length.[13] Black Cottonwood is the largest riparian tree species in the Columbia Plateau Ecoregion and the largest deciduous species in western North America. The species has a typical lifespan of 100 to 200 years (Braatne et al. 1996), and can attain a dbh of about 30 to 35 in (76 to 90 cm) in the Pacific Northwest (Franklin and Dyness 1988), but sizes are smaller in the Columbia Plateau. The interior of British Columbia is less productive than the province's coastal ecoregion, and in the interior, Black Cottonwood attain an average size of 15 in (38 cm) dbh and 52 ft (16 m) tall at age 80 years, and 26 in (66 cm) dbh and 70 ft (21 m) tall at age 150 years (Roe 1958). Other tree species that might also

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[13] Large wood is also known as large woody debris (LWD) or coarse woody debris. There is no universal definition of LWD. Another common definition is greater than 4 in (10 cm) in diameter and greater than 3 ft (1 m) in length. Small wood is also known as small woody debris or fine woody debris.

Large trees in riparian areas are a major source of instream large woody debris/Jim Cummins , WDFW

function as large wood are Quaking Aspen, White Alder, and Peachleaf Willow *Salix amygdaloides*. Daubenmire (1970) reported that Quaking Aspen and White Alder grow to about 8 and 23 in (10 and 58 cm) dbh, respectively, in riparian areas of the Columbia Plateau.

Substantial amounts of small wood may be recruited to and stored in stream channels through Beaver activity. Most dryland riparian areas lack large trees, but in dryland riparian areas Beaver create large roughness elements from small wood that mimic some effects of large wood (see section 7.3). In effect, Beaver convert small wood to large wood. Beaver dams consist mainly of mud and small wood. Published information on small wood sizes in dams of North American Beaver is lacking. However, Beaver most often forage on trees ranging from 1.2 to 3.2 in (3 to 8 cm) in diameter (Collen and Gibson 2001), and a large proportion of woody stems used as food are also used in dam construction (Barnes and Mallik 1996). In Maryland, Blersch and Kangas (2014) found that 98% of sticks (i.e., small wood) in a Beaver dam were less than 4 in (10 cm) in diameter and that 46% of those sticks were probably placed in the dam by Beaver. The other 54% of sticks in the dam were due to passive capture of transported wood.

Large wood recruitment processes in riparian areas of the Columbia Plateau should be very similar to those in forested ecoregions. Therefore, the main processes should be tree mortality caused by disease or senescence, bank erosion, and wind throw. Maximum recruitment distances for large wood will generally be either riparian zone width or site-potential tree height, whichever is smaller. I found no studies on recruitment mechanisms of small wood, nevertheless, the main mechanisms should be direct fall of branches into a stream channel, mobilization by overbank flooding, and Beavers. Consequently, depending on the relative distribution of small and large wood sources within a riparian area, the recruitment distance for small wood could be greater than the distance for large wood. That is, shrubs that lie farther from the stream channel than trees, could contribute small wood via flooding or Beaver.

7.4.3. Detrital Nutrients

With respect to riparian ecosystems, *detrital nutrients* most often refer to nutrients derived from terrestrial litter. Litter may consist of leaves, bark, seeds, cones, flowers, fruit, nuts, twigs, and other small plant parts (Benfield 1997). Terrestrial litter in an aquatic system is also called allochthonous nutrients.[14] Detrital nutrients are only one component of nutrient dynamics in riparian ecosystems. Riparian areas also mediate bi-directional exchange of dissolved nutrients between terrestrial and aquatic systems. Detrital nutrients in the larger context of nutrient dynamics (e.g., nitrogen cycle) are discussed in Chapter 6.

Besides research on marine derived nutrients (Naiman et al. 2002), the dynamics of macronutrients—nitrogen, phosphorus, and carbon (N, P, and C)—in riparian areas of the Pacific Northwest has been largely neglected by scientists (but see Naiman and Sedell 1980; Triska et al. 1984). I found one study related to nutrient dynamics in riparian areas that was conducted in the Columbia Plateau Ecoregion (e.g., Cushing and Wolf 1982; Cushing and Wolf 1984; Cushing 1988).

The availability of macronutrients has profound effects on the productivity, composition, and structure of ecosystems. In Washington, many streams and rivers in undisturbed watersheds are oligotrophic, i.e., exhibit low productivity associated with low N or P concentrations. Thut and Haydu reported in 1971 that roughly half of unpolluted surface waters in Washington were N-limited, and the other half were limited by P. Interestingly, N-limited waters, such as those in the Yakima River and lower Columbia River basins, most often arise from volcanic rock formations, while P-limited waters, such as those in the Okanogan River basin, flow from glacial deposits or granitic formations (Thut and Haydu 1971; Murphy 1998). Nutrient-limited waters are often sensitive to anthropogenic inputs of macronutrients, and when more N or P enters a stream than can be immediately metabolized, degradation of aquatic habitats often results. Artificial fertilizers, which are inorganic nutrients, cannot mimic the foodweb role of detrital nutrients. Artificial fertilizers are used by plants, but litter feeds heterotrophs such as macroinvertebrate detritivores (shredders and collectors), many of which are prey for salmonids. Therefore, providing adequate amounts of litter to streams is an important issue for riparian area management.

According to Xiong and Nilsson (1997), in drier and less productive grasslands, a certain amount of litter may help to conserve soil moisture and thus increase plant growth. The same effect may be obtained in dryland riparian areas. Hence, in addition to providing nutrients, litter from riparian vegetation may also retain soil moisture.

Little is known about nutrient dynamics in dryland riparian areas. Nitrogen cycling, for instance, is driven by microbes in soils, and microbial activity depends on water availability. In forested ecoregions, nitrogen cycling is rarely limited by water availability. Hence, microbial processes are active in both uplands and riparian soils, and can occur year round. In arid regions, nitrogen cycling in uplands may cease when soil moisture approaches zero. This is also true of nitrogen cycling in riparian areas, however, nitrogen dynamics are more complex in riparian areas because they are closely linked to seasonal water table elevations which are affected by site and watershed-scale hydrology (Belnap et al. 2005).

Particulate and dissolved organic carbon are important sources of energy in aquatic ecosystems. Both support organisms near the bottom of the food web. In forested ecoregions, productivity of headwater streams is typically light-limited by a closed canopy (Conners and Naiman 1984; Murphy 1998), and consequently, the main source of energy is allochthonous chemical energy

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[14] Allochthonous means originating or formed in a place other than where found. In contrast, autochthonous means originating or formed in the place where found.

from riparian areas. In arid and semi-arid regions, the canopy over headwater streams tends to be more open, which allows more light to reach the aquatic ecosystem. Cushing and Wolf (1982) found that chemical energy for a small spring-fed stream in central Washington, Rattlesnake Springs, was predominantly autochthonous, i.e., produced within the stream by Watercress *Rorippa nasturtium-aquatica,*[15] Cattail *Typha latifolia*, and algae. Direct allochthonous inputs contributed 11% of total energy inputs, and all of it was consumed. Consequently, Cushing and Wolf (1982) concluded that the aquatic community exhibited both autotrophic and heterotrophic traits.

The amount of allochthonous carbon produced by dryland riparian areas varies considerably. The input rates of allochthonous organic matter (specifically, leaf litter from shrubs) in two dryland riparian areas were 1.1 $oz/ft^2/yr$ (350 $g/m^2/yr$) at Rattlesnake Springs and 0.54 $oz/ft^2/yr$ (165 $g/m^2/yr$) at Snively Springs (Fisher 1995). The former is about 1.2 to 2.1 times greater than the amount of leaf litter produced by riparian areas in mesic forests (Gregory et al. 1991; cited in Fisher 1995).

Nearly all N in lotic ecosystems in forested ecoregions of the Pacific Northwest is derived from allochthonous sources. Headwater streams in the western Cascades of Oregon, for instance, obtain more than 90% of annual nitrogen inputs from biotic processes in the adjacent forest (Triska et al. 1984). An important source of N in forests of western Oregon and Washington is N_2-fixing Red Alder *Alnus rubra*, which contributes both litter and dissolved organic N to streams (Compton et al. 2003). Small streams in the Columbia Plateau Ecoregion must also obtain N from allochthonous sources. The overstory of some riparian plant communities in that ecoregion consists of N_2-fixing White Alder *Alnus rhombifolia* or Thinleaf Alder *Alnus incana*. Evidence suggests that Black Cottonwood and willow species contain mutualistic organisms that fix nitrogen (Doty et al. 2009, Wuehlissch 2011). This helps to explain how these species can survive and grow on recently disturbed, barren, floodplain sediments. Litter from these pioneering plants also provides nitrogen for other plant species in subsequent successional stages and deliver allochthonous nitrogen to the aquatic ecosystem.

The only study I could find on litter source distances was conducted in mature conifer forest in western Washington. Bilby and Heffner (2016) found that about 95% of litter delivered to streams comes from distances between 39 and 54% of mean tree height, depending upon forest type, topography, and wind. For Snively Springs, a spring-fed small stream with a riparian area comprised of dense willow and wild rose in central Washington, Cushing (1988) found that roughly 40% of litter fell directly into the stream from overhanging vegetation and 60% was blown laterally from plants some distance from the streambank. He did not measure distances. These proportions were affected by factors that influence wind, such as vegetation density and topography.

7.4.4. Shade

For an in-depth discussion of shade and water temperature refer to Chapter 4.

Numerous studies in dryland riparian areas have shown that salmonid species abundance is negatively correlated with stream temperature (Platts and Nelson 1989; Li et al. 1994; Ebersole et al. 2003a; Zoellick 2004). Even when primary and secondary production increase in response to reduced shading, salmonid populations often respond negatively to high temperatures (Tait et al. 1994). Shade is important because it directly affects stream temperatures. Other factors can significantly affect stream temperature, such as topographic shading, channel azimuth, channel form, groundwater flow, and hyporheic exchange, but for the purposes of riparian area management, I focus on shade provided by vegetation. Vegetation also affects local air temperature,

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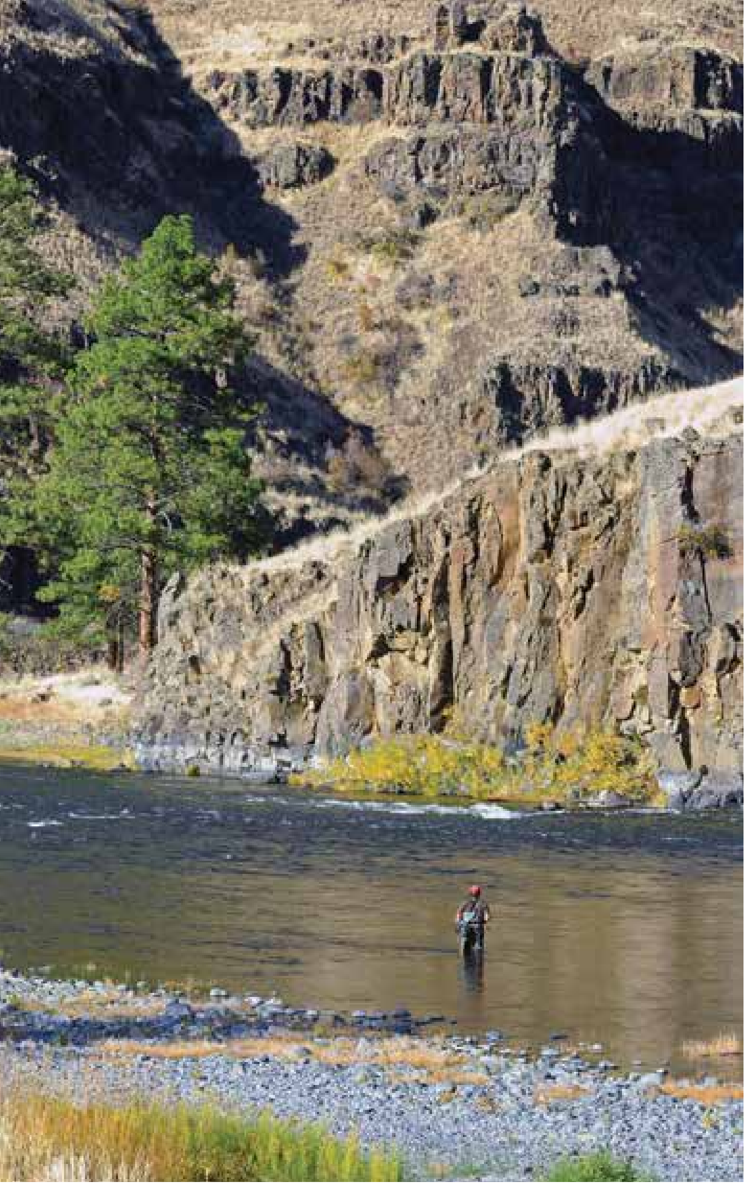
[15] Watercress (*Rorippa nasturtium-aquaticum*) is an exotic plant in Washington State.



In the Columbia Plateau, some stream reaches have no shade from vegetation/Alan Bauer

humidity, and air movement, which all affect stream temperature (Cristea and Janisch 2007).

Shading of a stream is affected by the vegetation's height and foliage density and by the stand's depth (or width) and stem density. In general, a taller, denser, or wider strip of vegetation will shade a larger proportion of a channel for a longer time, at both daily and annual time scales. Riparian vegetation also enables the formation of undercut banks, which provide shade along streambanks that has been shown to lower water temperatures of individual pools (Ebersole et al. 2003a; Ebersole et al. 2003b).

The classification system based on overstory underscores the substantial differences in shading provided by riparian vegetation types: tall tree, short tree, tall shrub, shrub, grass-like, grass, and forb (Crawford 2003). The obvious implication of these vegetation types is that some stream channels in the Columbia Plateau receive little or no shade from vegetation, even historically. However, the actual historical condition of many riparian areas is unknown. Studies show that restoration of riparian areas, especially restoration of incised channels, can alter vegetation type, increase shading, and reduce water temperatures. How spatial variation in vegetation type and consequent shading currently affects salmonid habitat regionally is also unknown. Ecologists lack a map of current and potential riparian vegetation types across the Columbia Plateau that would enable managers to assess impacts on fish-bearing streams from insolation (see Macfarlane et al. 2017 for a first approximation).

7.4.5. Pollutant Removal

For an in-depth discussion of pollutant removal by riparian areas refer to Chapter 5.

The capability of riparian areas to remove certain pollutants from runoff is well documented (Barling and Moore 1994; Hickey and Doran 2004; Polyakov et al. 2005; Dosskey et al. 2010). Management of riparian areas for pollutant removal differs from the other ecological functions discussed in this document in that the primary focus is on mitigating activities occurring outside the riparian area. The pollutant removal function is unique in that it only exists in the presence of human activities that generate polluted runoff, and it is only necessary when runoff from upland activities threatens to degrade in-stream water quality. Major nonpoint-source pollutants removed with riparian "buffers"[16] are sediments, excess nutrients (nitrogen or phosphorus), pesticides, and pathogens.

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[16] Riparian buffers are also known as riparian management zones (RMZs). "Buffer" refers to its purpose, which is to reduce or prevent adverse impacts to water quality, fisheries, and aquatic biodiversity from human activities occurring upslope of the buffer. Riparian buffers managed specifically for pollutant removal may also be called vegetated filter strips.

The processes in riparian areas affecting pollutant transport and fate are complex and often interrelated. Riparian areas can slow surface runoff and increase infiltration of water into the soil, thereby enhancing both deposition of solids and filtration of water-borne pollutants. Riparian areas also intercept and act on contaminants in subsurface flow through dilution, sorption, physical transformation, chemical degradation by various biogeochemical processes, and through uptake and assimilation by plants, fungi, and microbes. Hydrology, soils, and vegetation all affect pollutant transport and fate, but for the purposes of riparian area management I focus on vegetation.

Research on pollutant removal by riparian buffers conducted in arid or semi-arid ecoregions is very limited (Hook 2003). I could locate only three such studies (Tate et al. 2000; Fasching and Bauder 2001; Hook 2003), but all three have significant shortcomings. Hook (2003), for instance, simulated sediment-laden run-off from rangeland in western Montana. He reported that 20 ft (6 m) buffers removed at least 94% of sediment regardless of vegetation type in the buffer, but whether these simulations are realistic is unknown because he did not compare his run-off flow volumes and sediment concentrations to values from actual rangeland. Fasching and Bauder (2001), simulated sediment-laden run-off from tilled, bare cropland in central Montana. They found that 40 ft (12 m) wide buffers removed 75 to 80% of sediment during a simulated 50-year 24-hour storm event. However, while the buffer was 40 ft (12 m) wide, the bare cropland was only 15 ft (3 m) wide, which is certainly an unrealistic size for cropland. Tate et al. (2000) measured nitrate-nitrogen, total phosphorus, and suspended solids in run-off from irrigated pastures in the Central Valley of California. They reported that 33 ft (10 m) wide buffers did not significantly reduce nitrate-nitrogen concentration, total phosphorus concentration, or total suspended solid load. However, the strength of their inferences was compromised by extreme variability. Some buffered plots yielded higher concentrations of all three pollutants than unbuffered plots, which they attributed to differences in soil and microtopography among plots.

Despite the large quantity of research conducted around the world, no widely accepted recommendations have emerged on minimum buffer widths needed to protect water quality. The lack of agreement amongst scientists is due, in part, to the surprising complexity of the biogeochemical processes that remove pollutants from surface and subsurface flows, the many different environmental conditions among research sites, the variety of methods used to study pollutant removal, and the shortcomings of some research (as described above). One way to cut through this variability and develop quantitative relationships between buffer width and pollutant removal efficacy is meta-analysis.[17]
I found five such meta-analyses in the literature (Mayer et al. 2007; Liu et al. 2008; Yuan et al. 2009; Zhang et al. 2010; Sweeney and Newbold 2014).

The number of studies in the five meta-analyses ranged from to 52 to 88. I could determine the geographic locations of all studies included in four of the meta-analyses; only one study was conducted in an arid or semi-arid region (i.e., Tate et al. 2000, cited in Zhang et al. 2010). Four meta-analyses analyzed sediment studies, three meta-analyses analyzed nitrogen studies, and 1 one meta-analysis (Zhang et al. 2010) analyzed phosphorus and pesticide studies. All meta-analyses show that: 1) removal efficacy increases as buffer width increases, 2) removal efficacy varies among pollutants, and 3) relationships between buffer width and removal efficacy follow a nonlinear law of diminishing returns. In other words, the marginal removal efficacy decreases as buffer width increases. Comparing the four pollutants, buffer width that resulted in 90% removal was narrowest for sediment and widest for nitrogen (surface and subsurface combined). See Chapter 5 for a complete summary of their results.

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[17] Meta-analysis refers to various approaches for drawing statistical inferences from many separate but similar experiments (Mann 1990).

Only one study from arid or semi-arid ecoregions was included in the meta-analyses. Therefore, actual removal efficacy versus buffer width relationships may be quite different for riparian buffers in the Columbia Plateau. Differences in soil, hydrology, and vegetation between arid or semi-arid ecoregions and other ecoregions are significant, and are likely to affect pollutant removal. Because soils in arid ecoregions with biological crusts may be less permeable (Warren 2014) and vegetation tends to be less dense, especially in the upland zone of influence, the meta-analyses may underestimate buffer widths needed for a given level of removal efficacy. This issue can only be resolved through research on pollutant removal by dryland riparian areas.

7.4.6. Alluvial Water Storage

Alluvial water storage, also known as bank storage, is an ecological function of riparian areas that is thought to be especially important for small streams in dryland ecoregions (Pollock et al. 2003). However, the idea that alluvium can supply enough water to maintain base flows is controversial because empirical data to support it are lacking (S. Wondzell, U.S. Forest Service, personal communication).

Alluvial water storage is thought to be part of a cyclic process. During annual floods, stream stage exceeds water table elevation, and surface waters flow into shallow alluvial aquifers (Winter et al. 1998). During the dry season, as stream stage declines, water flow reverses from aquifers to surface water, and the released water may comprise a substantial portion of the stream's base flow (Brunke and Gonser 1997; Whiting and Pomerants 1997). By absorbing some floodwaters, alluvial aquifers may also reduce the impacts of flood events (Whiting and Pomerants 1997). The water volume in bank storage depends on the duration of flooding, the area of inundated floodplain, hydraulic gradients between surface water and the aquifer, soil infiltration rate, the permeability and porosity of alluvial sediments, floodplain width, and thickness of the bank storage layer (Whiting and Pomerants 1997; Chen and Chen 2003).

Autumn foliage, brown grass, and low flows annually coincide in Columbia Plateau Ecoregion/Scott Fitkin, WDFW

Alluvial fans, such as these emanating from side canyons, store water from spring snow melt that contributes to a river's base flow during the dry season/Scott Fitkin, WDFW

On the semi-arid plains of Alberta, Meyboom (1961) found that groundwater released from bank storage is a substantial portion of dry-season stream flow in the Elbow River. Bank storage was 73% of total groundwater discharge during normal base-flow recession, and during late summer, discharge from bank storage comprised 50 to 70% of the river's daily mean discharge. On the eastern edge of the Great Plains in central Iowa, Kunkle (1968) determined that base flow characteristics of Four Mile Creek were related mainly to the adjacent alluvial sand reservoir. Bank storage discharge accounted for 37 to 42% of the annual base flow.

Hydrological studies in the Yakima River basin (Synder and Stanford 2001, and studies cited therein) have suggested that prior to irrigation projects extensive exchange between surface and ground waters occurred in riparian areas and floodplains. Annual inundation of floodplains, including riparian areas, would recharge shallow surficial aquifers, including the alluvial aquifer. Synder and Stanford (2001) asserted for the Yakima River, "Based on fundamental hydrologic principles, there is no question that groundwater recharge of this nature would not only have maintained base flows, but would have provided areas of cooler thermal refugia." A salient characteristic of alluvial water storage is residence time, which can vary from days to years, depending upon floodplain width and hydraulic connectivity of alluvial sediments (Whiting and Pomereants 1997). The historical hydrology of rivers in the Columbia Plateau was water entered alluvial aquifers during spring peak flows and exited the aquifer weeks later during the low flows of late summer and fall of that same year (Synder and Stanford 2001). The historical timing and magnitudes of aquifer recharge and discharge may have been altered by dams, diversions, and levees.

Riparian and floodplain vegetation affect alluvial water storage in four ways. First, vegetation may reduce water storage through evapotranspiration. In fact, removing riparian vegetation can increase annual water yield from a watershed (Salemi et al. 2012). However, the timing of the additional surface water may not coincide with annual low flows when it is most needed. Second, through root growth and decay, vegetation creates soil macropores that increase soil permeability (Fisher and Binkley 2000) and thus enhance infiltration of floodwaters. Third, vegetation increases the residence time of floodwaters in floodplains by increasing surface roughness (Thomas and Nisbet 2007). The increased residence time provides more time for floodwaters to percolate through soils and into the alluvial aquifer. Vegetation increases surface roughness through stems, organic litter, large woody debris, and by altering micro-topography. Fourth, riparian vegetation helps to maintain cooler temperatures of shallow groundwater. Heat conduction from the soil surface into groundwater affects groundwater temperatures. Consequently, shallow groundwater temperatures are cooler in winter and warmer in summer. During the summer, shade from riparian vegetation and the riparian microclimate, which is cooler than adjacent uplands, may result in cooler groundwater. Stanford et al. (2002), for example, collected data suggesting that intact riparian forest moderated the rate of increase in groundwater temperatures along several reaches of the Yakima River. Alluvial aquifers typically underlie floodplains (Vaccaro 2011), including riparian areas. The alluvial water storage function of floodplains has been degraded by dams that reduce flood magnitudes, levees that reduce the area of flooding, and the channelization or loss of distributary channels across alluvial fans. To replace natural recharge pathways, water managers are trying artificial recharge of shallow alluvial aquifers (USBR & WDOE 2012; WWBWC 2013). While artificial recharge may replace the aquifer recharge function of floods, it cannot replace the fluvial disturbances caused by flooding that are needed to maintain riparian vegetation and aquatic habitats.

7.5. Conclusions

The goal for riparian areas of the Columbia Plateau Ecoregion is the same as the goal for forested ecoregions—maintain or restore key ecological functions. However, management to achieve that goal is more complicated in dryland riparian areas for three reasons. First, there is a greater diversity of plant communities within riparian ecosystems of the Columbia Plateau than in the surrounding forested ecoregions—the vegetation heights of dryland riparian ecosystems range from sedges to tall trees. Several key ecological functions of riparian areas—namely, shade, wood, and detrital nutrients for aquatic habitats—are dependent on vegetation height. The other three functions—bank stability, pollutant removal, and alluvial water storage—are largely dependent on processes occurring at or below the soil surface. In forested ecoregions, the total capacity of a riparian area to provide five of the key functions typically occurs within a site-potential tree height (FEMAT 1993). In other words, with respect to aquatic habitats, full function for five of the key riparian ecological functions is typically provided by an area that is one site-potential tree height wide.[18]

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[18] Site-potential" is a concept related to forest productivity. A site's productivity is often expressed as the height a particular tree species growing on that site is expected to attain by a specified age. FEMAT (1993) defined site-potential tree height as the average maximum height of the tallest dominant trees (200 years old or greater).

In dryland ecoregions, the width of the riparian ecosystem is based on the ecological function that creates the widest zone of influence. At most sites, this will be either the wood (large and small) or pollutant removal functions. For grass, herb, shrub, and small tree vegetation types, the zone of influence based on wood, which depends on vegetation height, will be narrower than the zone of influence based on pollutant removal. Therefore, if pollutants, including sediments due to ground disturbance, are a concern at a particular site, then riparian ecosystem width should be based on the desired removal efficacy for pollutants created at that site. If, for instance, surface run-off containing excess nitrogen is a concern and a 95% removal is desired, then the zone of influence is 170 ft (52 m) wide, measured from the active channel, see Table 5.4. If pollutant removal is not a concern, then riparian ecosystem width should be based on site-potential vegetation height (i.e., trees or shrubs), which should provide maximum wood and shade for aquatic habitats.

Some trees (Pinus ponderosa) *have all the luck/Scott Fitkin, WDFW*

The second reason management of riparian ecosystems is more complicated in the Columbia Plateau Ecoregion is water. The existence of dryland riparian areas depends on soil moisture and water table elevations. Many dryland riparian plant communities evolved under an annual hydrological cycle of flooding followed by gradual recession of stream flows. This cycle has been disrupted by dams and water diversions. The consequences of water management on riparian areas were documented by Braatne et al. (2007) who found an unsustainable age-class distribution of cottonwoods along the Yakima River. Two long-term impacts for these reaches of the Yakima could be reductions in shade and large wood for aquatic habitats. Water management is likely to have caused adverse changes to riparian plant communities along other rivers and streams in the Columbia Plateau. Under state law (RCW 90.03), in-stream flows are managed only to maintain flows above a specified minimum flow. Little consideration is given to mimicking the natural seasonal hydrograph for the sake of riparian ecosystems. To restore riparian ecosystems and aquatic habitats in the Columbia Plateau, that may need to change.

Third, management will be more complicated because many riparian areas in the Columbia Plateau have been highly altered by human activities, and ecologists do not know their historical or "properly functioning" conditions. Lacking such information will hamper the success of site-scale riparian restoration projects and regional restoration plans. An initial step toward grappling with this issue might be a subregional or WRIA-level mapping of potential riparian vegetation types. The mapping could include the capacity to support Beaver (e.g., Macfarlane et al. 2017) and the vegetation that would develop in the presence of Beaver. The map would serve three purposes: 1) a vegetation guide for riparian restoration projects, 2) a historical baseline for fish habitat conditions in the Columbia Plateau, and 3) habitat restoration objectives for the recovery of salmon and other aquatic species.

Historical information on riparian areas will serve three other purposes. First, knowledge of historical conditions will enable an accounting of lost ecosystem services. The losses incurred by fisheries are well recognized, but losses of wildlife, both game and nongame species, due to altered riparian areas have not been rigorously evaluated. Another major ecosystem service is the contribution to perennial stream flow from short-term storage of water in shallow alluvial aquifers that lie beneath floodplains (USBR & WDOE 2012; WWBWC 2013). Losses of this ecosystem service caused by disconnecting floodplains from streams also have not been rigorously evaluated. Second, knowledge of historical conditions may enable an estimate of the past quantity and quality of salmon habitats. Historical knowledge of past salmon habitats can be used in salmon recovery plans to establish habitat restoration objectives and priorities. The third purpose is partly political. Knowledge of past habitat loss can influence policy decisions to conserve remaining habitats or to restore habitats where practicable.

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7.7. Literature Cited[19]

Adam, S. 2004. Appendix E *in* B.E. Jamison and M.F. Livingston. Sage-Grouse habitat evaluation in the shrub-steppe ecosystem of the Yakama Reservation. Project final report. Wildlife, Range & Vegetation Resources Management Program, Confederated Tribes and Bands of the Yakama Nation, Toppenish, Washington. (vii)

AP (Associated Press). 2012. Umatilla tribes hope to trim wild horse numbers without slaughtering any. Available: **http://www.oregonlive.com/pacific-northwest-news/index.ssf/2012/10/umatilla_tribes_hope_to_trim_w.html**. (February 2018). (vii)

Anderson, E.W. 1987. Riparian area definition – a viewpoint. Rangelands 9(2):70. (i)

Bailey, R.G. 1995. Description of the ecoregions of the United States (2nd ed.). Misc. Pub. No. 1391. U.S. Forest Service, Washington, D.C. (viii)

Baker, B.W., H.C. Ducharme, D.C.S. Mitchell, T.R. Stanley, and H.R. Peinetti. 2005. Interaction of Beaver and Elk herbivory reduces standing crop of willow. Ecological Applications 15:110–118. (i)

Barling, R.D., and I.D. Moore. 1994. Role of buffer strips in management of waterway pollution: a review. Environmental Management 18:543-558. (i)

Barnes, D.M., and A.U. Mallik. 1996. Use of woody plants in construction of Beaver dams in northern Ontario. Canadian Journal of Zoology 74:1781-1786. (i)

Beechie, T.J., M.M. Pollock, and S. Baker. 2008. Channel incision, evolution and potential recovery in the Walla Walla and Tucannon River basins, northwestern USA. Earth Surface Processes and Landforms 33:784–800. (i)

Belnap, J., J.R. Welter, N.B. Grimm, N.N. Barger, and J.A. Ludwig. 2005. Linkages between microbial and hydrologic processes in arid and semi-arid watersheds. Ecology 86:298–307. (i)

Belsky, A.J., A. Matzkem, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419-431. (i)

Benfield, E.F. 1997. Comparisons of litterfall input to streams. Journal of the North American Benthological Society 16:104-108. (i)

Beschta, R.L., and W.J. Ripple. 2016. Riparian vegetation recovery in Yellowstone: the first two decades after Wolf reintroduction. Biological Conservation 198:93-103. (i)

Bilby, R.E., and J.T. Heffner. 2016. Factors influencing litter delivery to streams. Forest Ecology and Management 369:29-37. (i)

Bisson, P.A., R.E. Bilby, M.D. Bryant, C.A. Dollof, G.B. Grette, R.A. House, M.L. Murphy, K.V. Koski, and J.R. Sedell. 1987. Large woody debris in forested streams in the Pacific Northwest: past, present, and future. Pages 143-190 *in* E.O. Salo and T.W. Cundy, editors. Streamside management: forestry and fishery interactions. Contribution No. 57. Institute of Forest Resources, University of Washington, Seattle. (i)

Black, A.E., E. Strand, P. Morgan, J.M. Scott, R.G. Wright, and E. Watson. 1998. Biodiversity and land-use history of the Palouse bioregion: pre-European to present. Pages 85-99 *in* T.D. Sisk, editor. Perspectives on the land use history of North America: a context for understanding our changing environment. Biological Science Report USGS/BRD/BSR-1998-0003. U.S. Geological Survey, Reston, Virginia. (viii)

Blersch, D.M., and P.C. Kangas. 2014. Signatures of self-assembly in size distributions of wood members in dam structures of *Castor Canadensis*. Global Ecology and Conservation 2:204-213. (i)

Bohn, C. 1986. Biological importance of streambank stability. Rangelands 8(2):55-56. (i)

Braatne, J.H., R. Jamieson, K.M. Gill, and S.B. Rood. 2007. Instream flows and the decline of riparian cottonwoods along the Yakima River, Washington, USA. River Research and Applications 23:247–267. (i)

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[19] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Braatne, J.H., S.B. Rood, and R.E. Heilman. 1996. Life history, ecology and conservation of riparian cottonwoods in North America. Pages 57-85 *in* R.F. Stettler, H.D. Bradshaw, P.E. Heilman, and T.M. Hinckley, editors. Biology of *Populus* and its implications for management and conservation. NRC Research Press, Ottawa, Ontario. (i)

Brunke. M., and T. Gonser. 1997. The ecological significance of exchange processes between rivers and groundwater. Freshwater Biology 37:1-33. (i)

Bryce, S.A., and J.M. Omernik. 1997. Level IV ecoregions of the Columbia Plateau Ecoregion of Oregon, Washington, and Idaho. Pages 11-22 *in* S.E. Clark and S.A. Bryce, editors. Hierarchical subdivisions of the Columbia Plateau and Blue Mountains ecoregions. PNW-GTR-395. U.S. Forest Service, Pacific Northwest Forest Station, Portland, Oregon. (viii)

Buckley, G.L. 1993. Desertification of the Camp Creek drainage in central Oregon. Yearbook of the Association of Pacific Coast Geographers 55:91-126. (i)

Campbell, A.G., and J.F. Franklin. 1979. Riparian vegetation in Oregon's western Cascade Mountains: composition, biomass, and autumn phenology. Bulletin No. 14. Coniferous Forest Biome Ecosystem Analysis Studies, University of Washington, Seattle. (i)

Carlson, N.K. 1940. The history of grazing and livestock development in the Pacific Northwest. Range Division, Region 9, Soil Conservation Service. (viii)

Case, T.L., and J.B. Kauffman. 1997. Wild ungulate influence on the recovery of willows, Black Cottonwood, and Thin-Leaf Alder following cessation of cattle grazing in northeastern Oregon. Northwest Science 71:115-126. (i)

Chen, X., and X. Chen. 2003. Stream water infiltration, bank storage, and storage zone changes due to stream fluctuations. Journal of Hydrology 280:246-264. (i)

Collen, P., and R.J. Gibson. 2001. The general ecology of beavers (*Castor* spp.), as related to their influence in stream ecosystems and riparian habitats, and the subsequent effects on fish – a review. Reviews in Fish Biology and Fisheries 10:439–461. (i)

Compton, J.E., M.R. Church, S.T. Larned, and W.E. Hogsett. 2003. Nitrogen export from forested watersheds of the Oregon Coast Range: the role of N_2-fixing Red Alder. Ecosystems 6:773–85. (i)

Conners, M.E., and R.J. Naiman. 1984. Particulate allochthonous inputs: Relationships with stream size in an undisturbed watershed. Canadian Journal of Fisheries and Aquatic Sciences 41:1473–1488. (i)

Cooke, H.A., and S. Zack. 2008. Influence of Beaver dam density on riparian areas and riparian birds in shrubsteppe of Wyoming. Western North American Naturalist 68:365-373. (i)

Corenblit, D., E. Tabacchi, J. Steiger, and A.M. Gurnell. 2007. Reciprocal interactions and adjustments between fluvial landforms and vegetation dynamics in river corridors: a review of complementary approaches. Earth-Science Reviews 84: 56-86. (i)

Crawford, R.C. 2003. Riparian vegetation classification of the Columbia Basin, Washington. Natural Heritage Program Report 2003-03. Washington Department Natural Resources. Olympia, Washington. (viii)

Cristea, N., and J. Janisch. 2007. Modeling the effects of riparian buffer width on effective shade and stream temperature. Ecology Publication No. 07-03-028. Environmental Assessment Program, Washington State Department of Ecology, Olympia, Washington. (viii)

Cushing, C.E. 1988. Allochthonous detritus inputs to a small, cold desert spring-stream. Vereinigung fur Theoretische und Angewandte Limnologie 23:1107-1113. (i)

Cushing, C.E., and E.G. Wolf. 1982. Organic energy budget of Rattlesnake Springs, Washington. American Midland Naturalist 107:404-407. (i)

Cushing, C.E., and E.G. Wolf. 1984. Primary production on Rattlesnake Springs, a cold desert spring-stream. Hydrobiologica 114:229-236. (i)

Daubenmire, R.F. 1942. An ecological study of the vegetation of southeastern Washington and adjacent Idaho. Ecological Monographs 12:53-79. (i)

Daubenmire, R.F. 1970. Steppe vegetation of Washington. Washington State Experimental Station Technical Bulletin 62. Washington State University Cooperative Extension, Pullman, Washington. (viii)

Denfeld, D.C. 2011. Fort Walla Walla. Essay 9649, HistoryLink.org. Available: **http://www.historylink.org/File/9649.** (January 2019). (viii)

Dosskey, M.G., P. Vidon, N.P. Gurwick, C.J. Allan, T.P. Duval, and R. Lowrance. 2010. The role of riparian vegetation in protecting and improving chemical water quality in streams. Journal of the American Water Resources Association 46:261-277. (i)

Doty, S.L., B. Oakley, G. Xin, J.W. Kang, G.L. Singleton, Z. Khan, A. Vajzovic, and J.T. Staley. 2009. Diazotrophic endophytes of native Black Cottonwood and willow. Symbiosis 47:23–33. (i)

Dremmer, R., and R.L. Beschta. 2008. Recent history (1988-2004) of Beaver dams along Bridge Creek in central Oregon. Northwest Science 82:309-318. (i)

Ebersole, J. L., W.J. Liss, and C.A. Frissell. 2003a. Thermal heterogeneity, stream channel morphology, and salmonid abundance in northeastern Oregon streams. Canadian Journal of Fisheries and Aquatic Sciences 60:1266-1280. (i)

Ebersole, J. L., W.J. Liss, and C.A. Frissell. 2003b. Cold water patches in warm streams: physicochemical characteristics and the influence of shading. Journal of the American Water Resources Association 39:355-368. (i)

Evans, S. 1989. Provisional riparian and aquatic wetland plant communities on the Columbia Plateau. Report prepared under contract No. C0089098. Report to Washington Department of Ecology. Natural Heritage Program, Washington Department of Natural Resources, Olympia, Washington. (viii)

Fasching, R.A., and J.W. Bauder. 2001. Evaluation of agricultural sediment load reductions using vegetative filter strips of cool season grasses. Water Environment Research 73:590-596.

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management: an ecological, economic, and social assessment. United States Department of Agriculture and United States Department of Interior, Portland, Oregon. (viii)

Finley, W.L. 1937. The Beaver – conserver of soil and water. Pages 295-297 *in* Transactions of the Second North American Wildlife Conference. American Wildlife Institute, Washington, D.C. (viii)

Fisher, S.G. 1995. Stream ecosystems of the western United States. Pages 61-87 *in* C.E. Cushing, K.W. Cummins, and G.W. Minshall, editors. River and stream ecosystems of the world. University of California Press, Berkley, California. (i)

Fisher, R.F., and D. Binkley. 2000. Ecology and management of forest soils. Wiley, New York.

Florsheim, J.L., J.F. Mount, and A. Chin. 2008. Bank erosion as a desirable attribute of rivers. BioScience 58:519-529. (i)

Fonda, R.W. 1974. Forest succession in relation to river terrace development in Olympic National Park, Washington. Ecology 55:927-942. (i)

Fox, M., and S. Bolton. 2007. A regional and geomorphic reference for quantities and volumes of instream wood in unmanaged forested basins of Washington State. North American Journal of Fisheries Management 27:342-359. (i)

Franklin, J.F., and C.T. Dyrness. 1988. Natural vegetation of Oregon and Washington. Oregon State University Press, Corvallis, Oregon. (i)

Galbraith, W.A., and E.W. Anderson. 1971. Grazing history of the Northwest. Journal of Rangeland Management 24:6-12. (i)

Gebauer, A.D., R. Brown, S. Schwab, C. Nezat, and C. McNeely. 2015. Effects of an invasive grass (*Phalaris Arundinacea*) on water availability in semi-arid riparian zones. Wetlands 36:59–72. (i)

Giblin, D. 2006. Russian Olive, *Elaeagnus angustifolia*. Pages 78–79 *in* P.D. Boersma, S.H. Reichard, and

A.V. van Buren, editors. Invasive species in the Pacific Northwest. University of Washington Press, Seattle. (i)

Gibson, P.P., and J.D. Olden. 2014. Ecology, management, and conservation implications of North American Beaver (*Castor canadensis*) in dryland streams. Aquatic Conservation: Marine and Freshwater Ecosystems 24:391-409. (i)

Gibson, P.P., J.D. Olden, and M.W. O'Neill. 2015. Beaver dams shift desert fish assemblages toward dominance by non-native species (Verde River, Arizona, USA). Ecology of Freshwater Fish 24:355–372. (i)

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones. Bioscience 41: 540–551. (i)

Griffin, E., J. Kean, K. Vincent, J. Smith, and J. Friedman. 2005. Modeling effects of bank friction and woody bank vegetation on channel flow and boundary shear stress in the Rio Puerco, New Mexico. Journal of Geophysical Research 110:F04023. (i)

Gurnell, A. 2014. Plants as river system engineers. Earth Surface Processes and Landforms 39:4-25. (i)

Hickey, M.B., and B. Doran. 2004. A review of efficiency of buffer strips for the maintenance and enhancement of riparian ecosystems. Water Quality Research Journal of Canada 39:311-317. (i)

Higgs, E., D.A. Falk, A. Guerrini, M. Hall, J. Harris, R.J. Hobbs, S.T. Jackson, J.M. Rhemtulla, and W. Throop. 2014. The changing role of history in restoration ecology. Frontiers in Ecology and the Environment 12:499–506. (i)

Hood, W.G., and R.J. Naiman. 2000. Vulnerability of riparian zones to invasion by exotic vascular plants. Plant Ecology 148:105-114. (i)

Hook, P.B. 2003. Sediment retention in rangeland riparian buffers. Journal of Environmental Quality 32:1130-1137. (i)

Hough-Snee, N., B.B. Roper, J.M. Wheaton, and R.L. Lokteff. 2015. Riparian vegetation communities of the American Pacific Northwest are tied to multi-scale environmental filters. River Research and Applications 31:1151-1165. (i)

Hultine, K.R., and S.E. Bush. 2011. Ecohydrological consequences of non-native riparian vegetation in the southwestern United States: A review from an ecophysiological perspective. Water Resources Research 47: W07542. (i)

Hupp, C.R., and W.R. Osterkamp. 1996. Riparian vegetation and fluvial geomorphic processes. Geomorphology 14:277-295. (i)

Johnson, D.R., and D.H. Chance. 1974. Presettlement overharvest of upper Columbia River Beaver populations. Canadian Journal of Zoology 52:1519-1521. (i)

Jones, A. 2000. Effects of cattle grazing on North American arid ecosystems: a quantitative review. Western North American Naturalist 60:155-164. (i)

Kaiser, V.G. 1961. Historical land use and erosion in the Palouse – a reappraisal. Northwest Science 35:139-153. (i)

Katz, G.L., and P.B. Shafroth. 2003. Biology, ecology and management of *Elaeagnus angustifolia* L. (Russian Olive) in western North America. Wetlands 23:763-777. (i)

Kauffman, J.B., R.L. Beschta, N. Otting, and D. Lytjen. 1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries 22:12-24. (i)

Kauffman, J.B., and W.C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. Journal of Range Management 37:430-438. (i)

Kunkle, G.R. 1968. A hydrogeologic study of the ground-water reservoirs contributing base runoff to Four Mile Creek, east-central Iowa. Geological Survey Water-Supply Paper 1839-O. U.S. Geological Survey, Washington, D.C. (i)

KWA. 2004. Crab Creek subbasin plan. KWA Ecological Sciences Inc., Duvall, Washington. (viii)

Latterell, J.J., and R.J. Naiman. 2007. Sources and dynamics of large logs in a temperate floodplain river. Ecological Applications 17:1127-1141. (i)

Lesica, P., and S. Miles. 1999. Russian olive invasion into cottonwood forests along a regulated river in north-central Montana. Canadian Journal of Botany 77: 1077-1083. (i)

Li, H.W., G.A. Lamberti, T.N. Pearsons, C.K. Tait, and J.L. Li. 1994. Cumulative effects of riparian disturbance along high desert trout streams of the John Day Basin, Oregon. Transactions of the American Fisheries Society 123:627-640. (i)

Lichvar, R.W., N.C. Melvin, M.L. Butterwick, and W.N. Kirchner. 2012. National wetland plant list indicator rating definitions. ERDC/CRREL TN-12-1. U.S. Army Corps of Engineers, Washington, D.C. (viii)

Liu, X., X. Zhan, and M. Zhang. 2008. Majors factors influencing the efficacy of vegetated buffers on sediment trapping: a review and analysis. Journal of Environmental Quality 37:1667-1674. (i)

Liquori, M., and C.R. Jackson. 2001. Channel response from shrub dominated riparian communities and associated effects on salmonid habitat. Journal of the American Water Resources Association 37:1639-1651. (i)

Loope, L.L., P.G. Sanchez, P.W. Tarr, W.L. Loope, and R.L. Anderson. 1988. Biological invasions of arid land nature reserves. Biological Conservation 44:95-118. (i)

Lorne, L. 1993. Marketing wildlife: the Hudson's Bay Company and the Pacific Northwest, 1821-1849. Forest & Conservation History 37(January):14-24. (i)

Macfarlane, W.W., J.T. Gilbert, M.L. Jensen, J.D. Gilbert, N. Hough-Snee, P.A. McHugh, J.M. Wheaton, and S.N. Bennett. 2017. Riparian vegetation as an indicator of riparian condition: detecting departures from historic condition across the North American West. Journal of Environmental Management 202:447-460. (i)

Macfarlane, W.W., J.M. Wheaton, N. Bouwes, M.L. Jensen, J.T. Gilbert, N. Hough-Snee, and J.A. Shivik. 2017. Modeling the capacity of riverscapes to support Beaver dams. Geomorphology 277:72-99. (i)

Mack, R.N. 1988. First comprehensive botanical survey of the Columbia Plateau, Washington: the Sandberg and Leiberg Expedition of 1893. Northwest Science 62:118-128. (i)

Mack, R.N., and J.N. Thompson. 1982. Evolution in steppe with few large, hooved mammals. American Naturalist 119:757-773. (i)

Mann, C. 1990. Meta-analysis in the breech. Science 249:476-480. (i)

Mapes, L.V. 2010. Northwest tribes seek solutions to unwanted horses. The Seattle Times. Available: https://www.seattletimes.com/seattle-news/northwest-tribes-seek-solutions-to-unwanted-horses/. (February 2018). (viii)

Martinez, A.E., and P.F. McDowell. 2016. Invasive Reed Canarygrass (*Phalaris arundinacea*) and native vegetation roughness. Invasive Plant Science and Management 9:12-21. (i)

Mayer, P.M., S.K. Reynolds, M.D. Marshall, D. McCutchen, and T.J. Canfield. 2007. Meta-analysis of nitrogen removal in riparian buffers. Journal of Environmental Quality 36:1172-1180. (i)

McAllister, L.S. 2008. Reconstructing historical riparian conditions for two river basins in eastern Oregon, USA. Environmental Management 42:412-425. (i)

McKee, A., G. LaRoi, and J.F. Franklin. 1982. Structure, composition, and reproductive behavior of terrace forests, South Fork Hoh River, Olympic National Park. Pages 22-29 *in* E.E. Starkey, J.F. Franklin, and J.W. Matthews, technical coordinators. Ecological Research in National Parks of the Pacific Northwest. Oregon State University Research Laboratory Publication, Corvallis, Oregon. (i)

Meinig, D.W. 1968. The great Columbia Plain: a historical geography. University of Washington Press, Seattle, Washington. (i)

Meinzer, O.E. 1923. Outline of ground-water hydrology. Water Supply Paper 494. U.S. Geological Survey, Washington, D.C. (i)

Meyboom, P. 1961. Estimating ground-water recharge from stream hydrographs. Journal of Geophysical Research 66:1203-1214. (i)

Micheli, E.R., and J.W. Kirchner. 2002. Effects of wet meadow riparian vegetation on streambank erosion. 2. Measurements of vegetated bank strength and consequences for failure mechanics. Earth Surface Processes and Landforms 27:687–697. (i)

Minckley, W.L., and J.N. Rinne. 1985. Large woody debris in hot-desert streams: an historical overview. Desert Plants 7(3):142-153. (i)

Murphy, M.L. 1998. Primary productivity. Pages 144-168 *in* R.J. Naiman and R.E. Bilby, editors. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Naiman, R.J., and R.E. Bilby, editors. 1998. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Naiman, R.J., R.E. Bilby, D.E. Schindler, and J.M. Helfield. 2002. Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. Ecosystems 5:399-417. (i)

Naiman, R.J., T.J. Beechie, L.E. Benda, D.R. Berg, P.A. Bisson, L.H. MacDonald, M.D. O'Conner, P.L. Olsen, and E.A. Steel. 1992. Fundamental elements of ecologically healthy watersheds in the Pacific Northwest Coastal Ecoregion. Pages 127-188 *in* R.J. Naiman, editor. Watershed management: balancing sustainability and environmental change. Springer, New York. (i)

Naiman, R.J., C.A. Johnston, and J.C. Kelley. 1988. Alteration of North American streams by Beaver. BioScience 38:753-762. (i)

Naiman, R.J., and J.R. Sedell. 1980. Relationships between metabolic parameters and stream order in Oregon. Canadian Journal of Fisheries and Aquatic Sciences 37:834-847. (i)

NTHC (Northwest Tribal Horse Coalition). no date. Managing excess feral horses in the Inland Northwest. Available: **https://www.ynwildlife.org/Wildhorsecoalition.php.** (February 2018). (viii)

Ohmart, R.D. 1996. Historical and present impacts of livestock grazing on fish and wildlife resources in western riparian habitats. Pages 245-279 in P.R. Krausman, editor. Rangeland wildlife. Society for Range Management, Denver, Colorado. (i)

Olson, D.M., E. Dinerstein, E.D. Wikramanayake, N.D. Burgess, G.V.N. Powell, E.C. Underwood, J.A. D'Amico, H.E. Strand, J.C. Morrison, C.J. Loucks, T.F. Allnutt, J.F. Lamoreux, T.H. Ricketts, I. Itoua, W.W. Wettengel, Y. Kura, P. Hedao, and K. Kassem. 2001. Terrestrial ecoregions of the world: A new map of life on Earth. BioScience 51:933-938. (i)

Omernik, J.M. 1987. Ecoregions of the conterminous United States. Annals of the Association of American Geographers 77(1):118-125. (i)

Omernik, J.M. and A.L. Gallant. 1986. Ecoregions of the Pacific Northwest. EPA/600/3-86/033. U.S. Environmental Protection Agency, Corvallis, Oregon. (i)

Ortolano, L.K., K. Cushing, and contributing authors. 2000. Grand Coulee Dam and the Columbia Basin Project, USA. World Commission on Dams, Cape Town, South Africa. (viii)

Osterkamp, W.R., and C.R. Hupp. 2010. Fluvial processes and vegetation – glimpses of the past, the present, and perhaps the future. Geomorphology 116:274-285. (i)

Ott, J. 2003. "Ruining" the rivers in the Snake country. Oregon Historical Quarterly 104(2):166-195. (i)

Pabst, R.J., and T.A. Spies. 1999. Structure and composition of unmanaged riparian forests in the coastal mountains of Oregon, USA. Canadian Journal of Forest Research 29:1557-1573. (i)

Parker, M., F.J. Wood, B.H. Smith, and R.G. Elder. 1985. Erosional downcutting in lower order riparian ecosystems: have historical changes been caused by the removal of Beaver? Pages 35-38 in R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, R.H. Hamre, editors. Riparian ecosystems and their management: reconciling conflicting uses. RMRS-GTR-RM-120. U.S. Forest Service, Rocky Mountain Forest and Range Experimental Station, Fort Collins, Colorado. (viii)

Patten, D.T. 1998. Riparian ecosystems of semi-arid North America: diversity and human impacts. Wetlands 18:498–512. (i)

Platts, W.S., and R.L. Nelson. 1989. Stream canopy and its relationship to salmonid biomass in the intermountain west. North American Journal of Fisheries Management 9:446-457. (i)

Pollen-Bankhead, N., and A. Simon. 2010. Hydrologic and hydraulic effects of riparian root networks on streambank stability: Is mechanical root-reinforcement the whole story? Geomorphology 116:353-362. (i)

Pollock, M.M., T.J. Beechie, and C.E. Jordan. 2007. Geomorphic changes upstream of Beaver dams in Bridge Creek, an incised stream channel in the interior Columbia River basin, eastern Oregon. Earth Surface Processes and Landforms 32:1174-1185. (i)

Pollock, M.M., T.J. Beechie, J.M. Wheaton, C.E. Jordan, N.Bouwes. N. Weber, and C. Volk. 2014. Using Beaver dams to restore incised stream ecosystems. Bioscience 64:279-290. (i)

Pollock, M.M., M. Heim, and D. Werner. 2003. Hydrological and geomorphic effects of Beaver dams and their influence on fishes. Pages 213-233 in S.V. Gregory, K.L. Boyer, and A.M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Bethesda, Maryland. (i)

Polvi, L.E., E.E. Wohl, and D.M. Merritt. 2014. Modeling the functional influence of vegetation type on streambank cohesion. Earth Surface Processes and Landforms 39:1245-1258. (i)

Polyakov, V., A. Fares, and M.H. Ryder. 2005. Precision riparian buffers for the control of nonpoint source pollutant loading into surface water: a review. Environmental Reviews 13:129-144. (i)

Reidy Liermann, C.A., J.D. Olden, T.J. Beechie, M.J. Kennard, P.B. Skidmore, C.P. Konrad, and H. Imaki. 2012. Hydrogeomorphic classification of Washington state rivers to support emerging environmental flow management strategies. River Research and Applications 28:1340–1358. (i)

Richardson, J.S., R.J. Naiman, F.J. Swanson, and D.E. Hibbs. 2005. Riparian communities associated with Pacific Northwest headwater streams: assemblages, processes, and uniqueness. Journal of the American Water Resources Association 41:935-947.

Robison, G.E., J. Runyon, and C. Andrus. 1999. Cooperative stream temperature monitoring: project completion report for 1994–1995. Forest Practices Technical Report 2, Oregon Department of Forestry, Salem, Oregon. (viii)

Rockie, W.A. 1939. Man's effects on the Palouse. The Geographical Review 29(1):34-45. (i)

Roe, A.L. 1958. Silvics of Black Cottonwood. Miscellaneous Publication 17. U.S. Forest Service, Intermountain Forest and Range Experiment Station, Ogden, Utah. (viii)

Rood, S.B., S.G. Bigelow, M.L. Polzin, K.M. Gill, and C.A. Coburn. 2015. Biological bank protection: trees are more effective than grasses at resisting erosion from major river floods. Ecohydrology 8:772-779. (i)

Salemi, L.F., J.D. Groppo, R. Trevisan, J.M. de Moraes, W. de Paula Lima, and L.A. Martinelli. 2012. Riparian vegetation and water yield: a synthesis. Journal of Hydrology 454–455:195–202. (i)

Salo, E.O., and T.W. Cundy, editors. 1987. Streamside management: forestry and fishery interactions. Contribution No. 57. Institute of Forest Resources, University of Washington, Seattle. (viii)

Scheffer, P.M. 1938. The Beaver as an upstream engineer. Soil Conservation 3:178-181. (i)

Sedell, J.R., F.H. Everest, and D.R. Gibbons. 1989. Streamside vegetation management for aquatic habitat. Pages 115-125 in Proceedings of the National Silviculture Workshop: silviculture for all resources. U.S. Forest Service, Washington, D.C. (i)

Sedell, J.R., and F.J. Swanson. 1984. Ecological characteristics of streams in old-growth forest of the Pacific Northwest. Pages 9-16 in W.R. Meehan, T.R. Merrell, and T.A. Hanley, editors. Fish and wildlife relationships in old-growth forests. American Institute of Fishery Research Biologists. (i)

Simon, A., and A.J.C. Collison. 2002. Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. Earth Surface Processes and Landforms 27:527–546. (i)

Simon, A., N. Pollen, and E. Langendoen. 2006. Influence of two woody riparian species on critical conditions for streambank stability: upper Truckee River, California. Journal of the American Water Resources Association 42:99-113. (i)

Sleeter, B.M. 2012. Columbia Plateau Ecoregion. Pages 299- 236 in B.M. Sleeter, T.S. Wilson, and W. Acevedo, editors. Status and trends of land change in the Western United States – 1973 to 2000: U.S. Geological Survey Professional Paper 1794–A. U.S. Geological Survey, Reston, Virginia.

Stannard, M., and W. Crowder. 2001. Biology, history, and suppression of Reed Canarygrass (Phalaris arundinacea L.). Technical Note Plant Materials No. 40. Natural Resources Conservation Service, Boise, Idaho. (viii)

Stanford, J.A., E.B. Snyder, M.N. Lorang, D.C. Whited, P.L. Matson, and J.L. Chaffin. 2002. The reaches project: ecological and geomorphic studies supporting normative flows in the Yakima River basin, Washington. Flathead Lake Biological Station, Polson, Montana. (viii)

Stoddard, J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. 2006. Setting expectations for the ecological condition of streams: the concept of reference condition. Ecological Applications 16:1267–1276. (i)

Snyder, E.B., and J.A. Stanford. 2001. Review and synthesis of river ecological studies in the Yakima River, Washington, with emphasis on flow and salmon habitat interactions. Flathead Lake Biological Station, Polson, Montana. (viii)

Sweeney, B.W., and J.D. Newbold. 2014. Streamside buffer width needed to protect stream water quality, habitat, and organisms: a literature review. Journal of the American Water Resources Association 50:560-584. (i)

Talabere, A.G. 2002. Influence of water temperature and Beaver ponds on Lahontan Cutthroat Trout in a high-desert stream, southeastern Oregon. Master's thesis. Oregon State University, Corvallis. (i)

Tait, C.K., J.L. Li, G.A. Lamberti, T.N. Pearsons, and H.W. Li. 1994. Relationships between riparian cover and the community structure of high desert streams. Journal of the North American Benthological Society 13:45-56. (i)

Tate, K., G. Nader, and D. Lewis. 2000. Evaluation of buffers to improve the quality of runoff from irrigated pastures. Journal of Soil and Water Conservation 55:473–478. (i)

Thomas, F.M. 2014. Ecology of phreatophytes. Progress in Botany 75:335-375. (i)

Thomas, J.W., C. Maser, and J. E. Rodiek. 1979. Wildlife habitats in managed rangelands – the Great Basin of southeastern Oregon: riparian zones. GTR-PNW-80. U.S. Forest Service, Pacific Northwest Experimental Station, Portland, Oregon. (viii)

Thomas, H., and T.R. Nisbet. 2007. An assessment of the impact of floodplain woodland on flood flows. Water and Environment Journal 21:114-126. (i)

Thut, R.N., and E.P. Haydu. 1971. Effects of forest chemicals on aquatic life. Pages 159-171 in J.T. Krygier and J.D. Hall, editors. A symposium on forest land uses and the stream environment. Continuing Education Publications, Oregon State University, Corvallis. (i)

Toledo, Z.O., and J.B Kauffman. 2001. Root biomass in relation to channel morphology of headwater streams. Journal of the American Water Resources Association 37:1653-1663. (i)

Tribal Tribune. 2013. Horses on the landscape. Available: http://www.tribaltribune.com/news/article_3abf8d87-e606-54b1-9e42-5ef6c0fb1036.html. (February 2018). (viii)

Triska, F.J., J.R. Sedell, K. Cromack Jr, S.V. Gregory, and F.M. McCorison. 1984. Nitrogen budget for a small coniferous forest stream. Ecological Monographs 54:119-40. (i)

Tu, M. 2006. Reed Canarygrass, *Phalaris arundinacea*. Pages 12–13 *in* P.D. Boersma, S.H. Reichard, and A.V. van Buren, editors. Invasive species in the Pacific Northwest. University of Washington Press, Seattle. (i)

Tuttle, G.M., G.L. Katz, J.M. Friedman, and A.P. Norton. 2016. Local environmental context conditions the impact of Russian Olive in a heterogeneous riparian ecosystem. Invasive Plant Science and Management 9:272-289. (i)

USACE (U.S. Army Corps of Engineers). 2002. Final Lower Snake River juvenile salmon migration feasibility report/environmental impact statement, Appendix I: Economics. Walla Walla District, Washington. (i)

USBR (U.S. Bureau of Reclamation). 2017a. Yakima Project. Available: **https://www.usbr.gov/projects/index.php?id=400**. (September 2018). (viii)

USBR (U.S. Bureau of Reclamation). 2017b. Columbia Basin Project. Available: **https://www.usbr.gov/projects/index.php?id=438**. (September 2018). (viii)

USBR & WDOE (U.S. Bureau of Reclamation and Washington Department of Ecology). 2012. Final programmatic environmental impact statement Yakima River basin integrated water resource management plan. Yakima, Washington. (viii)

USFS (U.S. Forest Service). 2004. Riparian restoration. 0423 1201–SDTDC. Technology & Development Program, San Dimas, California. (viii)

USFWS (U.S. Fish and Wildlife Service). 1994. An ecosystem approach to fish and wildlife conservation. Washington, D.C. (viii)

Vaccaro, J.J. 2011. River-aquifer exchanges in the Yakima River basin, Washington. U.S. Geological Survey, Reston, Virginia. (viii)

Warren, S.D. 2014. Role of biological soil crusts in desert hydrology and geomorphology: implications for military training operations. Reviews in Engineering Geology 22:177-186. (i)

Webb, R.H., and S.A. Leake. 2006. Ground-water and surface-water interactions and long-term change in riverine riparian vegetation in the southwestern United States. Journal of Hydrology 320:302-323. (i)

Weddell, B.J. 2001. Changing perspectives in nineteenth century written descriptions of Palouse and canyon grasslands. Technical Bulletin No. 01-13. Bureau of Land Management, Cottonwood District, Cottonwood, Idaho. (viii)

West, E., and G. Ruark. 2004. A long, long time ago: historical evidence of riparian forests in the Great Plains and how that knowledge can aid with restoration and management. Journal of Soil and Water Conservation 59:105A-110A. (i)

Westbrook, C.J., D.J. Cooper, and B.W. Baker. 2006. Beaver dams and overbank floods influence groundwater-surface water interactions of a Rocky Mountain riparian area. Water Resources Journal 42:W06404. (i)

Westbrook, C.J., D.J. Cooper, and B.W. Baker. 2011. Beaver assisted river valley formation. River Research and Applications 27:247–256. (i)

White, P., and J.L. Walker. 1997. Approximating nature's variation: selecting and using reference information in restoration ecology. Restoration Ecology 5:338-349. (i)

Whiting, P.J., and M. Pomerants. 1997. A numerical study of bank storage and its contribution to streamflow. Journal of Hydrology 202:121-136. (i)

Winter, T.C., J.W. Harvey, O.L. Franke, and W.M. Alley. 1998. Ground water and surface water: a single resource. Circular 1139. U.S. Geological Survey, Denver, Colorado. (viii)

Wissmar, R.C. 2004. Riparian corridors of eastern Oregon and Washington: functions and sustainability along lowland-arid to mountain gradients. Aquatic Sciences 66:373-387. (i)

Wissmar, R.C., J.E. Smith, B.A. McIntosh, H.W. Li, and G.H. Reeves. 1994. A history of resource use and disturbance in riverine basins of eastern Oregon and Washington (early 1800s-1990s). Northwest Science 68 (special issue):1-35. (i)

Wohl, E. 2005. Compromised rivers: understanding historical human impacts on rivers in the context of restoration. Ecology and Society 10(2):2. (i)

Wohl, E., B.P. Bledsoe, R.B. Jacobson, N.L. Poff, S.L. Rathburn, D.M. Walters, and A.C. Wilcox. 2015. The natural sediment regime in rivers: broadening the foundation for ecosystem management. Bioscience 65:358-371. (i)

WSDA (Washington State Department of Agriculture). 2016. Washington State noxious weed general distribution maps. Olympia, Washington. Available: **https://agr.wa.gov/washington-agriculture/maps/weed-distribution**. (January 2020). (viii)

Wuehlisch, G.V. 2011. Evidence for nitrogen-fixation in the Salicaceae family. Tree Planter Notes 54(2):38-41. (i)

WWBWC (Walla Walla Basin Watershed Council). 2013. Walla Walla basin aquifer recharge strategic plan. Milton-Freewater, Oregon. (viii)

Xiong, S., and C. Nilsson. 1997. Dynamics of leaf litter accumulation and its effects on riparian vegetation: a review. Botanical review 63:240–264. (i)

YN (Yakama Nation). 2010. Wild Horse (K'u-See) Project. Wildlife, Range & Vegetation Resources Management Program, Confederated Tribes and Bands of the Yakama Nation, Toppenish, Washington. Available: **https://www.ynwildlife.org/wildhorseprogram.php.** (February 2018) (viii)

YSPB (Yakima Subbasin Planning Board). 2004. Yakima subbasin plan. Yakima Subbasin Fish and Wildlife Planning Board, Benton and Yakima counties, Washington. (viii)

Yuan, Y., R.L. Bingner, and M.A. Locke. 2009. A review of effectiveness of vegetative buffers on sediment trapping in agricultural areas. Ecohydrology 2:321-336. (i)

Zhang, X., X. Liu, M. Zhang, and R.A. Dahlgren. 2010. A review of vegetated buffers and a meta-analysis of their mitigation efficacy in reducing nonpoint source pollution. Journal of Environmental Quality 39:76-84. (i)

Zoellick, B.W. 2004. Density and biomass of redband trout relative to stream shading and temperature in southwestern Idaho. Western North American Naturalist 64:18-26. (i)

Chapter 8. Watersheds

By: *Derek B. Booth*

8.1. Introduction

This chapter on watershed-scale processes and management looks beyond the typical site-by-site perspective at which riparian management and stream protection is normally applied. This chapter also considers the interactions between site-specific treatments, whose collective effects may be greater (or less) than anticipated by the individual actions themselves, and it uses these insights to offer a broader approach to riparian management for the protection of aquatic systems. Given this expanded focus, the conditions and influences discussed here may lie outside the framework of normal guidance and regulatory mandates, but they nonetheless affect the health and productivity of riparian areas and their associated aquatic systems. Thus, the conceptual framework provided here should be useful when considering the likely effects of riparian management policies or practices.

The purpose of this chapter is to explore the relationship (and the importance) of watershed-scale management to achieving the goal of abundant, self-sustaining fish and wildlife populations. It assumes that a broader watershed-scale perspective on aquatic ecosystems will ultimately enhance the beneficial effects of management applied at all scales.

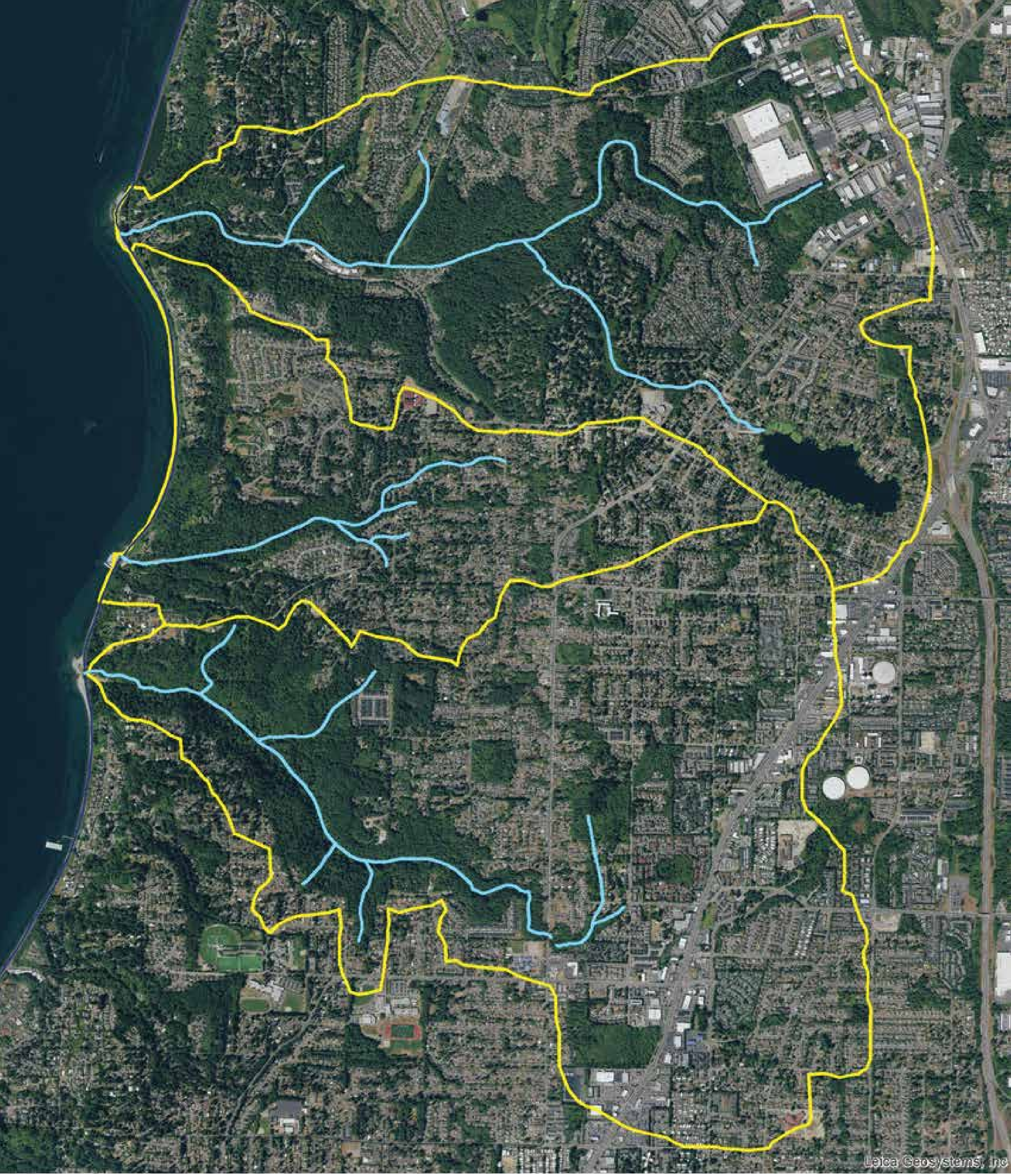
The chapter is divided into four primary sections:

1. A conceptual framework that identifies the key watershed processes and their primary influences on riparian ecosystems and their watercourses, a perspective with a long history in the scientific literature. However, advances in the discipline of landscape ecology over the past two decades—particularly the recognized importance of interrelationships between landscape elements in both spatial (e.g., connectivity and discontinuities) and temporal (e.g., disturbances and recovery) domains—have changed how biological and physical scientists view aquatic ecosystems and the riverine landscape that they occupy.

2. A discussion of how riparian management at the site scale, the emphasis of the other chapters in this document, can best support ecological functions and improve habitat conditions for fish and wildlife, even if a complementary set of watershed-scale management actions (e.g., land use at the watershed scale) cannot be implemented.

3. A description of how individual site-scale management actions, and their recognized site-scale benefits, relate to one another across a channel network and/or landscape. In other words, how do the consequences and overall value/benefit of isolated actions integrate along a channel network? Are the effects of individual actions simply additive, or are there synergistic effects that are revealed by a landscape ecology perspective?

4. Guidance, developed from an understanding of watershed processes and the perspective of landscape ecology, for improving the management of riparian ecosystems. This goal is complicated by the fact that most watersheds are managed under diverse and often uncoordinated responsibility: different landowners and managers have different goals and objectives. Furthermore, many of these managers have neither responsibility nor a process by which to coordinate their



Three watersheds (outlined in yellow) in urban areas of north Seattle/Terry Johnson, WDFW actions, especially over the long durations necessary for riparian management actions to achieve their greatest effects. This chapter offers no prescription for resolving these challenges, but it proceeds under the belief that greater awareness of these interactions will ultimately lead to more successful protection of riparian systems, streams, and their biota.

8.2. Conceptual Framework

This chapter seeks to integrate the functions of riparian ecosystems into the overall landscape in which they interact—the riverine landscape or *riverscape*. This riverine landscape differs from the contributing watershed of the river or stream: although the flux of water and water-borne biota respect the drainage divides of a watershed almost exactly, other components do not. In particular, mobile organisms (e.g., birds, and large mammals such as American Beaver *Castor canadensis*, and bear (*Ursus* spp.), wind-borne seeds, and wildfire all have the potential to affect and be affected by landscape elements both within and beyond the boundaries of any given watershed, and so they must be included in any comprehensive conceptual framework (Figure 8.1). Despite these watershed-crossing elements, the lotic (flowing-water) environment itself is strongly organized by the unidirectional flow of water, a property that is distinct from terrestrial environments, and so rivers commonly display systematic patterns in the downstream direction (e.g., the "river continuum" of Vannote 1980.) They also display systematic cross-stream patterns, notably the spatial, physical, and ecological relationships between the channel and its adjacent floodplain; and to a less visible but still significant degree in the vertical dimension as well, particularly the interaction of shallow groundwater with stream water in the hyporheic zone (Boulton et al. 1998).

Landscape ecology is the scientific discipline best suited to this larger perspective of the riverine landscape: "...we perceive a need to conceptualize rivers not as sampling points, lines, or gradients, but as spatially continuous longitudinal and lateral mosaics. As such, heterogeneity in the river landscape, or riverscape, becomes the focus of study" (Fausch et al. 2002:485) (Figure 8.2). The foundational premise of landscape ecology is that both biotic and abiotic elements of the landscape interact with one another, and those interactions are spatially mediated (Malanson 1995).

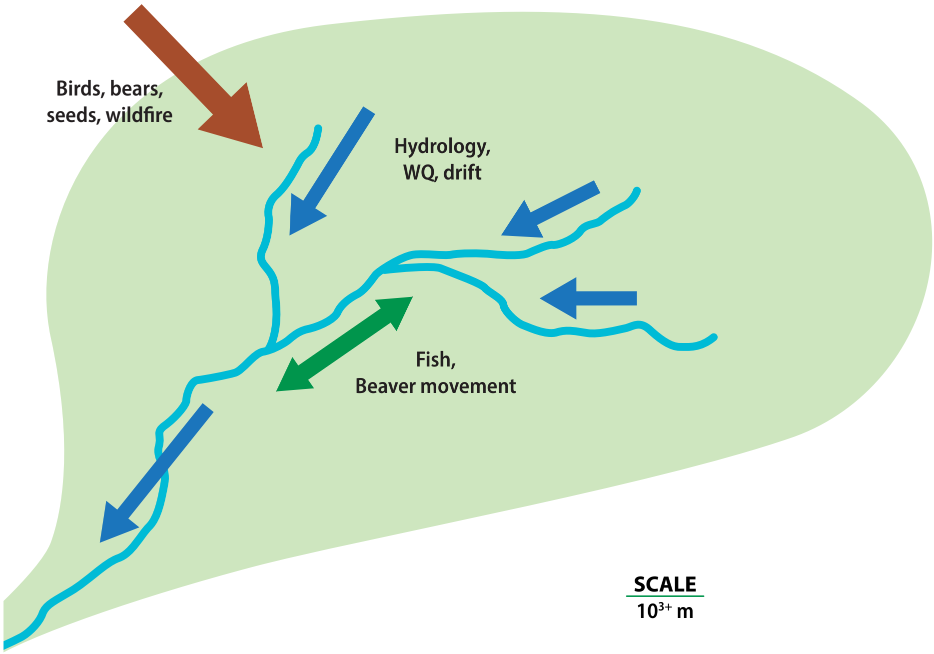


Figure 8.1. Movement of water and biota at the whole-watershed scale. WQ refers to water quality, and drift refers to the downstream transport of bottom-dwelling invertebrates by flowing water.

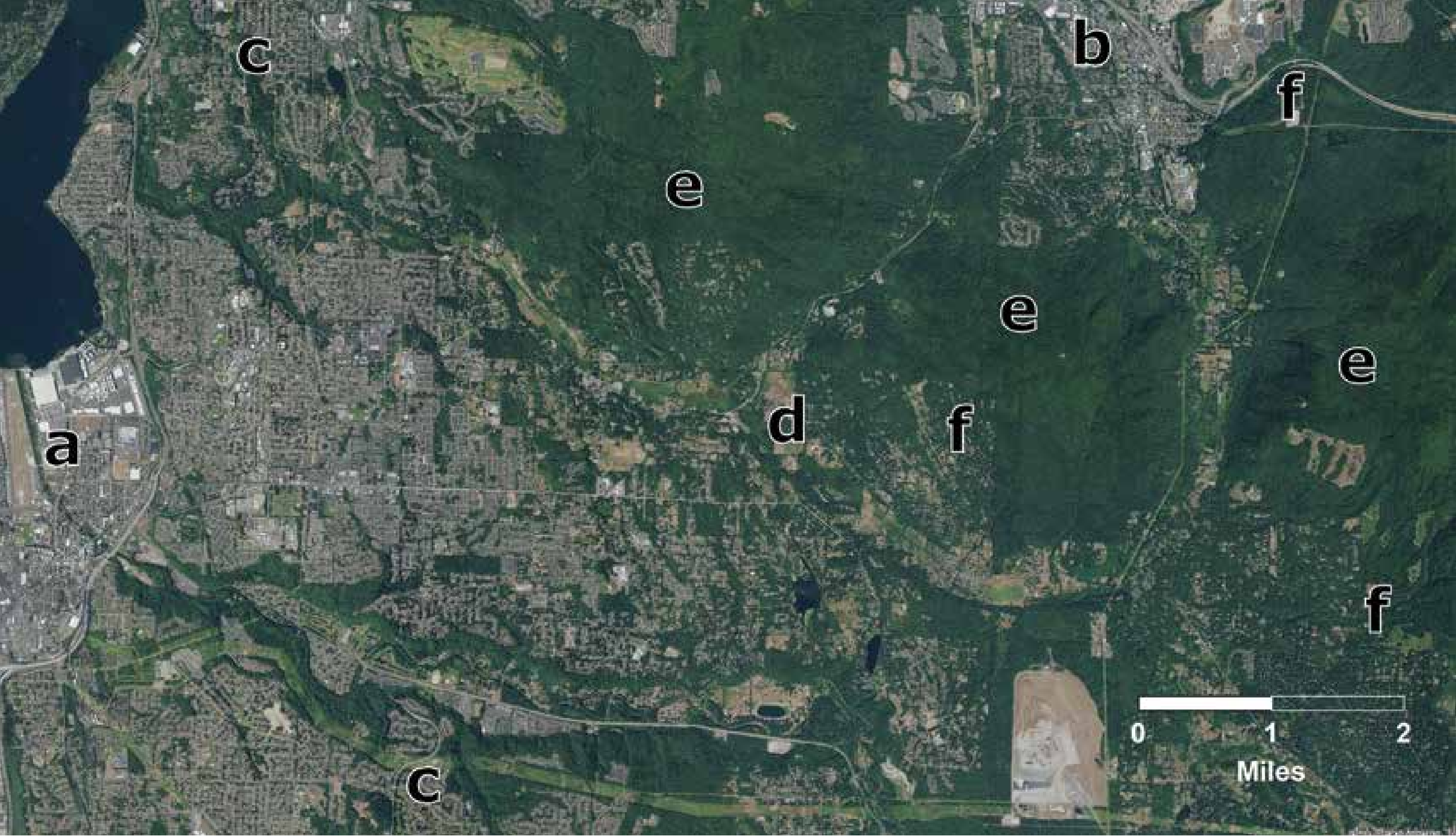


Figure 8.2. An example of landscape heterogeneity at multiple scales in an aerial view that includes the urban centers of Renton (west edge, labeled a) and Issaquah (northeast corner, b); suburban development throughout the south-central and northwest parts of the image (c); agriculture and rural-residential development, particularly in the valleys of May Creek and McDonald Creek trending NW across the center of the image (d); and extensive forest lands of Cougar, Squak, and Tiger Mountains (e) in the north-central and east parts. At finer spatial scales within the largely undeveloped areas in the eastern part of the image, fragmentation is nonetheless widespread from roads, utility lines, and clearing for residential lots and subdivisions (f).

This perspective emphasizes the importance of spatial heterogeneity; it also acknowledges the key role of episodic disturbances in determining the flux of energy and materials, the movement of organisms, and the creation of unique dynamics within the watershed (Wiens 2002).

At a finer spatial scale, water and biota interact across the permeable and dynamic boundary between riparian areas and a watershed's uplands (Figure 8.3, left panel). These areas comprise multiple components. Within riparian areas, these components include surface waters, in-channel bars and islands, floodplains, marshes, wet meadows, alluvial forests, and woody debris. For the uplands, these components might include unstable slopes, aquifer recharge areas, wetlands, seeps, forests, and areas of intensive human modification. Both areas also include the vast variety of organisms that occupy one or more of these habitats.

River running through rural residential area/Jane Atha, WDFW

Riparian areas occupy a particularly dynamic and disturbance-prone region (Figure 8.3, right panel) that can experience decadal, annual, or even more frequent flood events that locally alter, rearrange, or obliterate its components altogether.

> *In the natural state, riverine landscapes exemplify* the new paradigm in ecology *(sensu Talbot 1996), in which ecological systems are widely recognized as non-deterministic, open systems in continual states of flux, rather than internally regulated, homeostatic systems exhibiting equilibrium conditions. Yet, despite their highly dynamic nature, riverine landscapes provide predictable ecological conditions. Although individual landscape components exhibit high turnover, largely as a function of interactions between fluvial dynamics and successional phenomena, their relative abundances in the river corridor as a whole tend to remain constant over ecological time. [From Ward et al. 2002:518.]*

So, for example, a flood may scour a gravel bar at one location in a river and deposit sediment in another; but along any given channel reach the relative frequency of pools and bars may not vary substantially over time. The resulting transience and diversity of habitats provide a variety of habitats that, although ideally always present in aggregate, are not necessarily static in place or time (Robinson et al. 2002). Thus, managers should recognize this dynamic equilibrium as the key attribute of a truly functional riparian–aquatic ecosystem and make the protection of its dynamic *behavior* (rather than any specific *feature*) the overarching goal of riparian management.

Existing policies, however, generally do not facilitate management for dynamic conditions:

> *Policies setting underlying riparian goals that are essentially static and homogenous have become an integral part of many of the management guidelines that drive the goals for riparian management on federal, state and private lands in western Washington. Discussions [at a statewide forest management symposium] did not indicate that current management and policy guidelines necessarily preclude practices that might produce dynamic and heterogeneous riparian conditions. Because they promote uniform conditions over large areas, however, current policy and management guides do little to encourage treatments that would lead to ecologically diverse landscapes that could maintain critical functions that were formerly produced by natural disturbance regimes... [From Ryan and Calhoun 2010:viii-ix].*

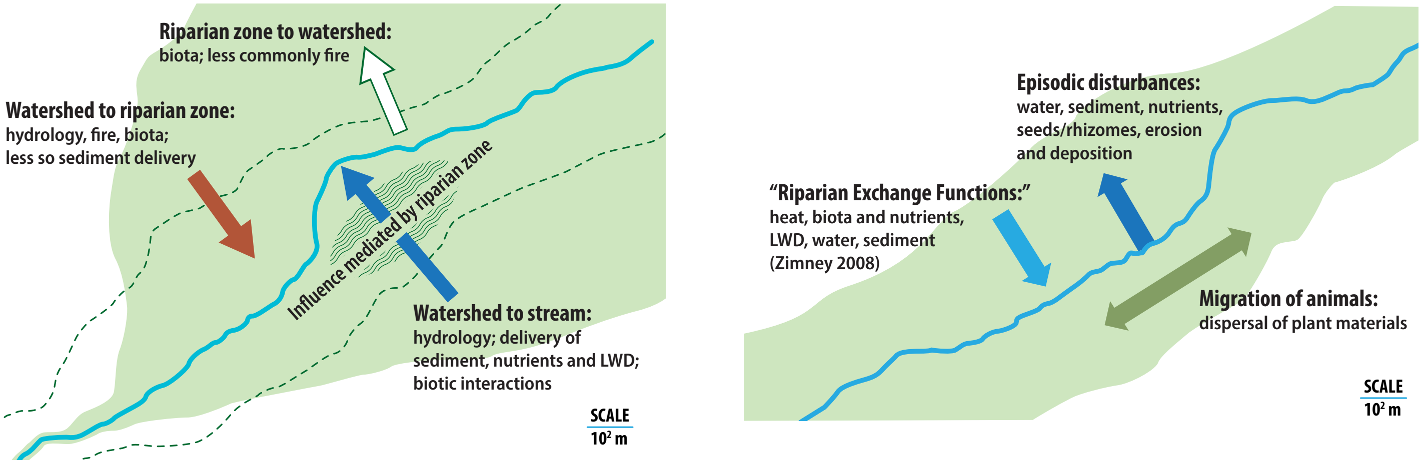


Figure 8.3. Movement of water, biota, and other elements of the aquatic system from the watershed into and through riparian areas (left) and between riparian areas and the stream (right).

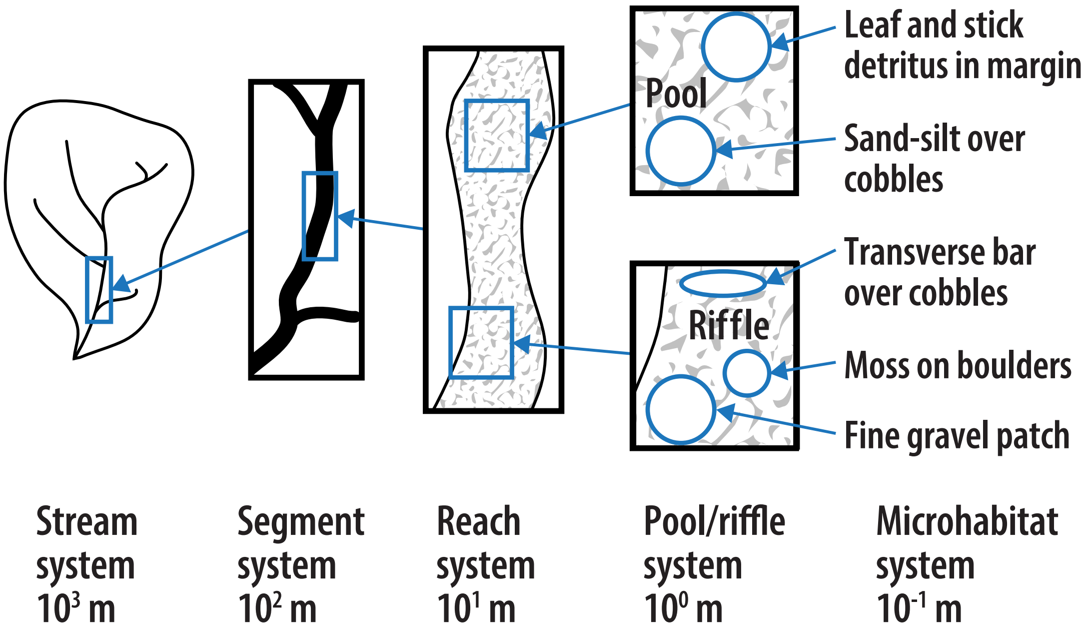


Figure 8.4. Hierarchical spatial organization of a watershed and its aquatic habitats. Approximate spatial scale, appropriate to second or third-order mountain stream (modified from Frissell et al. 1986).

On the other hand, riparian areas have experienced high levels of anthropogenic disturbance, both directly (e.g., harvest, road building) and indirectly (e.g., windthrow, landslides, and forest fire) due to forest management, which also compromises the dynamic equilibrium in riparian-aquatic ecosystems. In response, policy makers have reduced forest management-related disturbances in riparian areas, with the intent of passively reestablishing underrepresented late-seral classes (such as mature forest) which should improve riparian ecological functions, and conserve fish and wildlife species dependent on older forest. Such policies are all likely necessary parts of a successful riparian management strategy. However, because they do not address watershed-scale processes, these policies by themselves are almost surely insufficient to achieve desired riparian and aquatic ecosystem outcomes.

8.2.1. What are Watershed Processes?

Although the immediate goal of riparian management is the protection of species and their habitats, Frissell et al. (1986:199) suggested that "[w]e begin with the assumption that structure, operation, and other aspects of the organization and development of stream communities are largely determined by the organization, structure, and dynamics of the physical stream habitat." (Using the terminology elsewhere in this document, these terms correspond to the structure, composition, and functions of physical habitat). If so, "the problem becomes one of understanding these physical patterns across time and space. This requires a broad, integrative framework that places streams, their habitats, and their communities in wider geographic context" (Frissell et al. 1986:200). Frissell et al. (1986) developed this wider context by describing a spatial hierarchy that recognizes the influence of larger scale conditions and processes acting primarily (but not exclusively) unidirectionally on smaller scales (Figure 8.4). Factors that are best characterized over the largest extents, and that exert their influence over longest time frames, can be thought of as the ultimate "drivers" of the conditions and processes at lower levels in the hierarchy.

Three overarching process drivers—climate, geology, and topography—structure the suite of landscape-forming processes that are distributed over a landscape and that, in turn, govern watershed characteristics and processes (paraphrasing the framework of Montgomery 1999). At management time scales (i.e., years to decades), these process drivers are invariant and immutable—they are the "givens" under which a watershed evolves. The physical habitat features of any watershed do not simply arise from these process drivers directly, however; instead, they are created and continuously modified by the fluxes of materials (particularly water and sediment) and energy that arise from these process drivers and are widely termed watershed processes. Watershed processes are defined as "[t]he dynamic physical and chemical interactions that form and maintain the landscape and ecosystems on a geographic scale of watershed to basins (i.e., hundreds to thousands of square miles)" (Stanley et al. 2011:8).

To this simple, unidirectional, physical model of watershed influences must be added a few complications of lesser but still critical importance. First, the flow of influence is not always one-way: although topography exerts the dominant influence on the flow of water, over time the erosion by that flowing water will alter the topography to some degree (which, over geologic time scales, may be sufficient to change the very boundaries of the watershed). Second, biota are not always passive respondents to the presence and changes of physical habitat, but instead can also influence that physical habitat in ways that are only slowly being recognized (see Moore 2006 for an overview of such effects). Robinson et al. (2002) emphasizes the role of fauna as ecosystem engineers of riverine landscapes, for which the modification of stream substrates by spawning salmon (Montgomery et al. 1996) or the rearrangement of floodplain dynamics by Beaver (Pollock et al. 2003) are two widely recognized examples. Biological conditions and processes are also of fundamental importance to the expression of some watershed processes: the flux of nutrients may not alter the physical habitat but is key to the health of the ecosystem; the character and extent of vegetative cover will greatly mediate the effects of topography, climate, and geology on the movement of runoff, the downslope transport of sediment, and the potential for dramatic, process-altering wildfire. Finally, the effects of people and human activities cannot be ignored in any useful conceptual framework applied to the modern landscape. Although direct modification of habitat and streams is an obvious impact with a long historical legacy in the Pacific Northwest (and beyond), the aggregate effects of human activities across whole watersheds can be even more consequential to instream resources, to the extent that land use, in many settings, becomes no less of a process driver (i.e., a determinant in a watershed) than is topography or climate.

These concepts are integrated into Figure 8.5, which emphasizes the unidirectional, hierarchical flow of influence from the large-scale process drivers onto the watershed processes that, in turn, give rise to the instream physical and ecological conditions and responses. This representation is not intended to deny the *upstream* influences that also occur (see Beechie et al. 2010, Figure 1 as an alternative representation), but it sacrifices completeness for clarity in acknowledging the overriding importance of this unidirectional perspective. This conceptual model also articulates the variety of processes that are embraced within each of the major watershed processes we

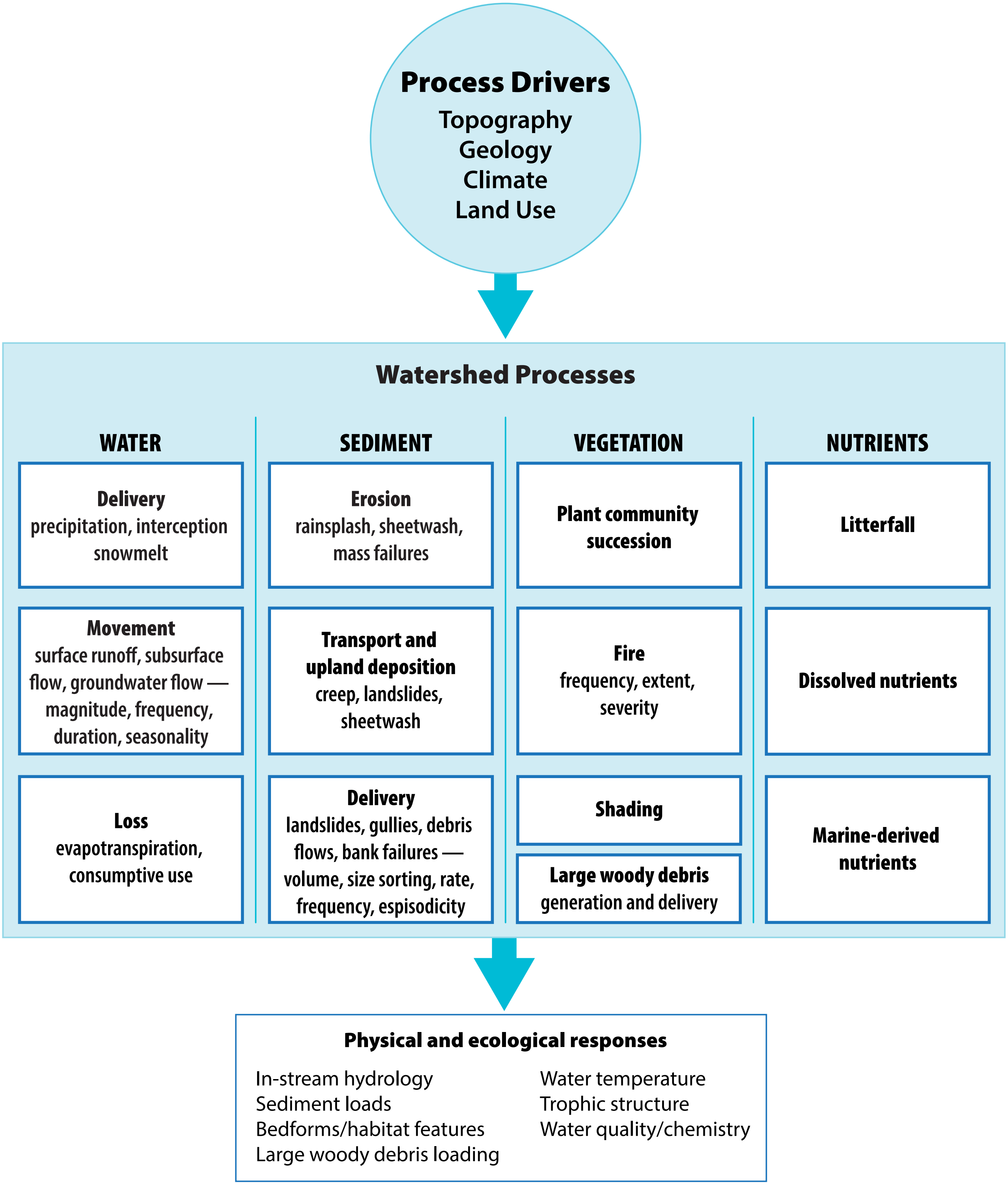


Figure 8.5. Conceptual framework for the hierarchical relationship of invariant, large-scale process drivers, the suite of watershed processes that are determined by these drivers, and the instream physical and biological responses to those processes. Multiple additional interactions between elements and levels are not shown on this diagram to emphasize the primary influences; but in any given setting, one or more of these secondary interactions may temporarily achieve equivalent importance. This framework embraces the definition of watershed processes from Stanley et al. (2011:8): "[t]he dynamic physical and chemical interactions that form and maintain the landscape and ecosystems."

recognize (water, sediment, vegetation, and nutrients), and it reminds us of the major instream responses to these dynamic watershed processes.

8.2.2. How Does the Watershed Affect Instream Conditions?

The direct effects (both physical and biological) of watershed processes on the stream channel are most commonly understood as the delivery of material from the watershed, with or without some attenuation or other modifications as they pass through the riparian ecosystem: water, sediment, nutrients, pollutants, and large wood. These materials are delivered primarily by surface water flow, subsurface flow, landslides and debris flows (Figure 8.6). These fluxes of material and energy from the broader riverine landscape interact with those arising from the riparian corridor itself, to different degrees and over different time scales, with a resulting complexity that defies our current efforts to quantitatively predict their expression in the physical form or biological condition of a stream.

The complexity of these interactions is highlighted by relatively weak correlations reported between various instream features and landscape predictors, regardless of the spatial scale being evaluated (e.g., whole-watershed vs. riparian areas). For example, Anlauf et al. (2011) evaluated the correlation between eleven instream habitat features with landscape variables characterizing both process drivers (e.g., watershed gradient, rainfall) and management-influenced factors (e.g., forest cover, disturbance history, road density) across 121 coastal Oregon streams. The strongest relationships showed unexceptional correlations between particular habitat features and the suite of landscape variables (active channel width, $R^2 = 0.72$; percentage of fine sediments, $R^2 = 0.48$). Other habitat features expressed essentially no relationship (percentage gravel, $R^2 = 0.05$; pools per 328 ft (100 m), $R^2 = 0.11$; percentage secondary channel area, $R^2 = 0.12$). Including management-influenced predictors increased adjusted R^2 values only modestly (by up to 16%). Many of the strongest relationships followed intuitive expectations, such as: “Wood volume was associated with several landscape predictors reflecting wood availability (% non-forest, % small trees, and % remnant forests), whereas pools per 328 ft (100 m) were associated with disturbances affecting pool retention (cow density, road density) and pool formation (% small trees)” (Anlauf et al. 2011:708). However, no relationship was sufficiently strong to suggest that either deterministic understanding or statistical correlations are sufficient to precisely characterize the influence of watershed-scale processes on a stream.

Our inability to predict instream conditions from watershed processes and riparian conditions with any useful degree of accuracy is widely recognized. In part, this inability is because the riparian corridor itself has significant, complex effects on the stream (see other chapters in this document). In addition, all of these influences—at every spatial scale—occur within a time-dependent framework mediated by rates of vegetation growth and decay, seasonal variations in flow, and episodic disturbances from floods, and even less frequent but potentially long-lasting channel impacts from fire or landslides (Lake 2000; Roper et al. 2007; Luce et al. 2012; Jellyman et al. 2013).

> *[E]mpirical associations between land use and stream response only varyingly succeed in implicating pathways of influence. This is the case for a number of reasons, including (a) covariation of anthropogenic and natural gradients in the landscape; (b) the existence of multiple, scale-dependent mechanisms; (c) nonlinear responses; and (d) the difficulties of separating present-day from historical influences. [From Allan 2004:257.]*

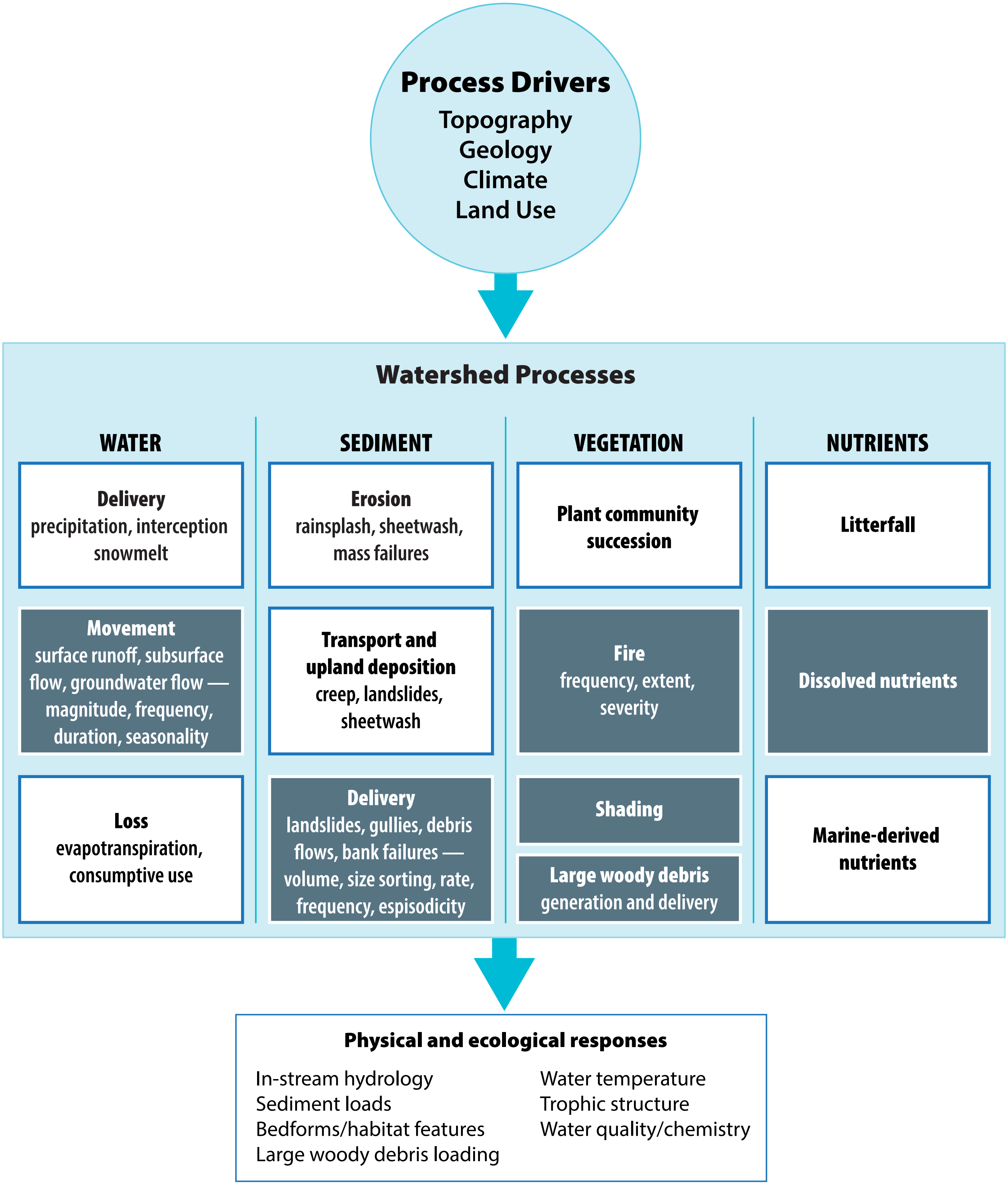


Figure 8.6. The conceptual framework of Figure 8.4, highlighting the subset of watershed processes (dark shaded boxes) that most directly affect instream conditions.

Clearcut harvest in a managed forest landscape/Jane Atha, WDFW

Regardless of our present inability to make reliable predictions of instream conditions from watershed processes, watershed processes do create and maintain instream habitat, and so the determinants of those processes are of indirect, but ultimately critical importance, to the understanding and the management of aquatic resources. Because the nature and intensity of these processes are strongly influenced by prevailing (and past) watershed land use(s), the effects of the dominant land use activities of the Pacific Northwest are discussed individually.

Forestry

The effects of watershed-scale forestry on instream conditions have been a primary focus of scientific research and forest practice regulations for many decades. Most of the attention has been paid to several discrete problems: increased stream temperature from loss of riparian vegetation, the degree to which large-scale loss of forest canopy alters the hydrologic regime, whether the creation of new roads has a discernible impact on both hydrology and sediment delivery to the channel network, and whether forest practices increase the frequency and intensity of debris flows and other forms of mass failure that reach the stream.

The potential for hydrologic alteration in logged watersheds focuses on two major mechanisms: 1) the loss of forest canopy, which should increase total water yield by reducing interception and evapotranspiration, and increase accumulation of snow on canopy-less ground, which increases the total volume of water available during subsequent rain-on-snow events, and 2) more rapid drainage of surface runoff to stream channels from a newly created road network. The review article of Andréassian (2004:12) provides a convenient summary of much of the voluminous literature on this topic, concluding that “deforestation could definitely increase both flood volumes and flood peaks. However, this effect is much more variable than the effect on total flow and may even be inverted in some years or in some seasons." The underlying causes for such increases are associated not only with the loss of forest canopy but also (and, perhaps, primarily) the consequences of soil compaction and development of the road network.

Increased sediment delivery to streams from watersheds is an anticipated outcome of both the direct and indirect effects of logging. Direct consequences result from the loss of canopy cover, increasing the potential for rainsplash and sheetwash erosion in areas of intrinsically low infiltration capacity, and the eventual loss of soil cohesion in areas of steep slopes. Indirect effects result from the construction of roads, for which inappropriate siting can result in sheetwash erosion and a greater incidence of landslides and debris flows from either concentrated road runoff or failure of the road prism itself. A variety of case studies making use of detailed field mapping, radionuclide tracers, and suspended sediment measurements have implicated various landscape features and erosive processes in post-logging watersheds, such as skid trails and landings, promoting surface runoff and erosion (Wallbrink et al. 2002), enhanced gully erosion (Reid et al. 2010), and road-initiated debris flows (May 2002).

Nutrient dynamics across the watershed are also affected by forest practices, with potential effects throughout the ecosystem (including the channel network and its riparian corridor). Richardson et al. (2005) offered a well-focused and detailed evaluation of the consequences of watershed disturbance, including logging, on the delivery and retention of organic matter in small streams of the Pacific Northwest. He noted that the replacement of coniferous with deciduous trees in recently logged areas can increase the delivery of nutrients, but that more rapid breakdown of delivered material and the accompanying reduction in both large wood and channel complexity, commonly expressed throughout the channel network of logged watersheds, "could produce a substantial reduction in stored organic matter." (Richardson et al. 2005:930).

Agriculture

Cultivated agriculture normally occupies a comparatively narrow range of low-gradient settings in a watershed. Thus, processes associated with steep slopes—particularly landslides and debris flows—found in logged watersheds are nearly absent here. However, the working of the agricultural land surface is far more intensive where it does occur, and a wider range of water-borne contaminants are potentially introduced. As with forestry, hydrology, sediment delivery, and nutrient dynamics are all affected by agricultural land use, but the nature and severity of its impacts on instream resources can be quite different. Runoff from agricultural watersheds, notably its load of nutrients and pesticides, were the historical impetus for the use of riparian buffers under an assumption of surface-runoff filtration (Osborne and Kovacic 1993). Brown et al. (2009) found that the legacy of prior agricultural land use, even in areas presently undergoing urbanization, is one of the strongest determinants of instream conditions in their nine study areas nationwide.

Grazing is a second, broad category of agricultural activities with significant impacts to streams, primarily as a result of direct encroachment into riparian areas and the channel itself. Belsky et al. (1999:2) reviewed dozens of mid- to late-20th century evaluations of grazing throughout arid lands of the American West that evaluated impacts to water quality and temperature, hydrology, channel geomorphology,

Watershed processes help create and maintain instream habitat. Thus the determinants of those processes are indirect but critically important to the understanding and the management of aquatic systems.

wildlife, and instream and riparian-zone vegetation. Their conclusion was that "[n]o positive environmental impacts were found" as a result of grazing, and they advocate complete cessation of grazing as the only sure course for recovery of riparian areas. More recent work (e.g., Agouridis et al. 2005; Raymond and Vondracek 2011) suggest that various strategies of rangeland management may have local success at reducing the magnitude of such impacts; but the large spatial extent over which grazing occurs in certain areas of the Pacific Northwest continues to make this a regionally significant impact to streams and riparian areas.

Urbanization

Urbanization is widely recognized to be the most intensive land use affecting watershed processes. It is also typically the most permanent, insofar as urban developments are rarely returned to anything approaching a natural state, a transition that is not precluded by agriculture and is quite practical in commercially managed forests. As a pervasive, near-total conversion of the land surface, urbanization has the ability to alter each of the dominant watershed processes wherever it occurs: hydrology (both surface and groundwater); the delivery of sediment, nutrients, pollutants, and sunlight; and the interaction of upland/terrestrial biota with the stream channel (Booth et al. 2004; Walsh et al. 2005). Nearly all of these alterations stem from the same fundamental change: the conversation of the land surface, with its vegetative cover and natural soils, to a variety of impervious and near-impervious surfaces. Undisturbed riparian corridors are widely recognized as useful protective measures for streams in urban areas, but all multi-scale evaluations of urban land cover and instream quality affirm the shared importance of not only riparian areas but also upland urbanization in ultimately determining instream conditions (e.g., Wang et al. 2001; Morley and Karr 2002). These watershed-level effects are significant at even low levels of urbanization (e.g., Vietz et al. 2014), and to date they have proven virtually impossible to control under typical riparian-management policies, highlighting the continuing challenge of achieving meaningful aquatic-system protection in an urbanizing watershed.

8.2.3. How Does the Watershed Affect Riparian Areas?

Changes to an upland watershed and watershed processes can also affect the conditions and functions of riparian areas directly (Figure 8.7). Although a riparian area is commonly defined in terms of its relationship to the stream, it also has a relationship to the uplands that surround it. Water from the stream channel will commonly be the riparian area's dominant source of both overland flow (during floods) and groundwater, but adjacent upland water sources can support riparian hydrology during periods of low flow. Much of the sediment that originates from the uplands is transported directly to the streams via channeled flow, but some fraction of that load will deposit before ever reaching the main channel, particularly if the riparian area includes a broad, low-gradient floodplain or if delivery occurs via episodic and channel-overtopping debris flows. Upland vegetation "buffers" riparian areas, providing

More important than simply protecting the habitat itself is protecting the processes that create and sustain that habitat. For this strategy to be successful, however, space in which that habitat can be expressed must also be protected.

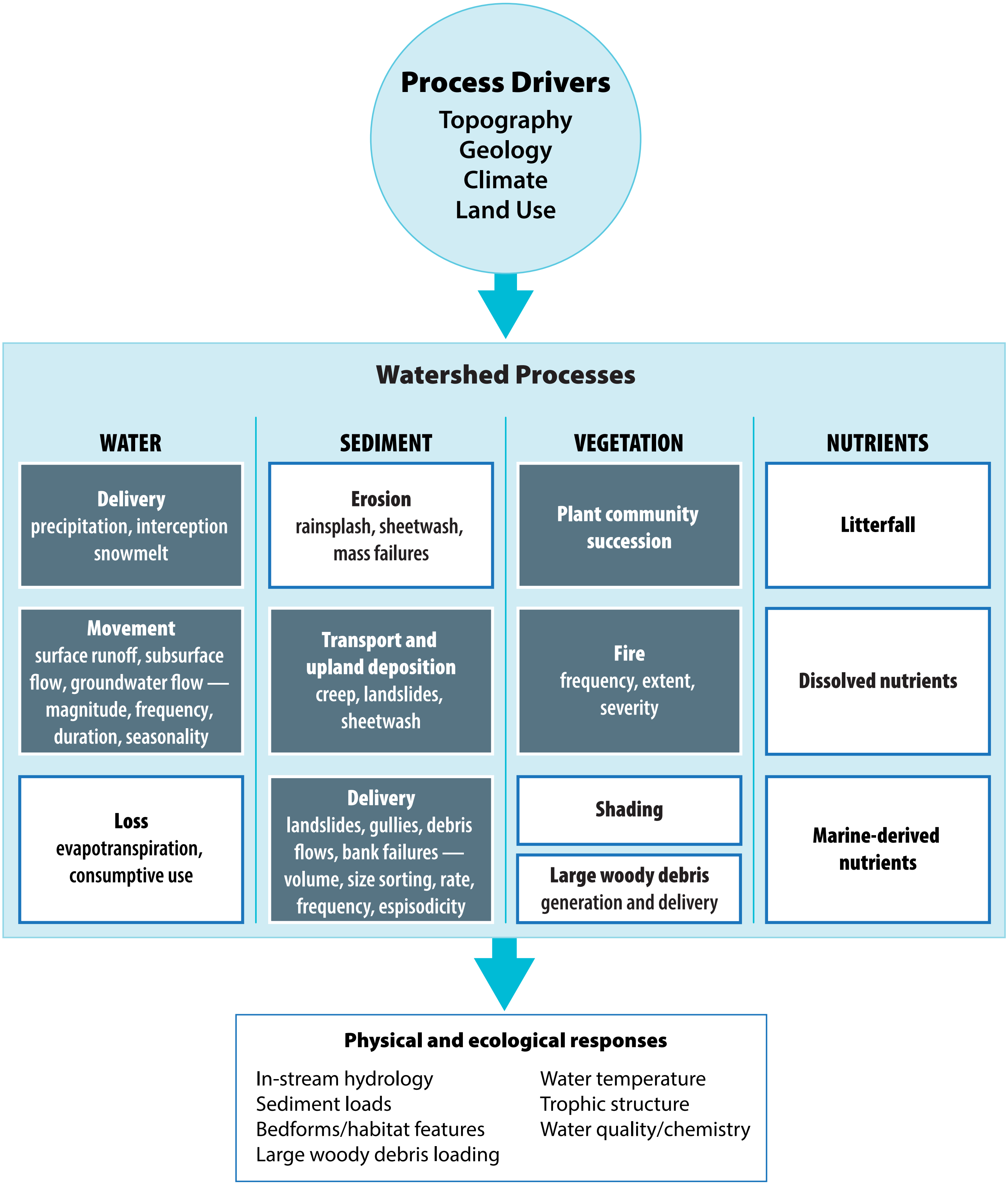


Figure 8.7. The conceptual framework of Figure 8.4, highlighting the subset of watershed processes (dark shaded boxes) that most directly affect conditions in riparian areas.

ecological connectivity between upland plant and animal communities with their riparian counterparts, and providing protection from mechanical disturbances, such as windthrow. Uplands also constitute a broader environment that either supports or suppresses wildfires that may freely cross the upland/riparian boundary.

Various land uses are relatively consistent with respect to their particular impacts to riparian ecosystems. That is, all three of the primary types (forestry, agriculture, urban) physically remove native vegetation, with differences only in the degree of impact. The loss of upland vegetation will consistently affect ecological connectivity and the risks of windthrow and fire regardless of the specific upland land use.

The interactions between uplands, riparian areas, and instream conditions are particularly complex with respect to wildfire. Arkle et al. (2010:299) found that "increasing riparian burn severity and extent were associated with greater year-to-year variation, rather than a perennial increase, in sediment loads, organic debris, large woody debris (LWD) and undercut bank structure." They emphasized a decrease in the stability of physical habitat and high variability in macroinvertebrate communities, which they attributed to annually fluctuating loadings of sediment, large wood and other organic material. They did not recognize any tendency for the macroinvertebrate community to become more like those of relatively undisturbed, reference streams over time, suggesting that changes to wildfire regime can result in permanent changes to stream ecosystems, and that these changes are a result not only of direct influence of burned areas on the stream but also of a permanently altered structure of the riparian corridor. With greater human disturbance, invasive species with high flammability (including those that have begun transforming riparian areas from fire breaks to fire-prone areas of a landscape; Coffman et al. 2010) become more common, as does the greater overall frequency of wildfire in modern human-dominated landscapes (Keeley et al. 2009). Thus this process driver, operating largely independent of typical considerations of riparian area management, can have a profound influence on instream resources (Pausas and Keeley 2014).

Wildfire in Methow River valley, 2018/ Scott Fitkin, WDFW

8.3. The Role and Limitations of Site-Scale Riparian Area Management

Whereas large-scale watershed processes govern the interactions and delivery of water, sediment, organic material, and nutrients to the stream over long distances, the effects of riparian vegetation on physical and ecological processes occur over much shorter distances with complex, direct, time-dependent interactions. These effects are expressed along river corridors "by influencing temperature and light regimes; producing organic detritus (leaf litter, woody debris); by routing water and sediment; by structuring the physical habitat at several scales; by providing a substrate for biological activity and habitat/cover for aquatic, amphibious and terrestrial animals" (Ward et al. 2002:524). For this reason, particular management attention to riparian areas is warranted, insofar as this part of a watershed has a unique, intimate interaction with the stream channel and its biota. Previous chapters explore these specific interactions in detail.

In addition to spatial relationships, temporal interactions are fundamental attributes of aquatic and riparian habitats. These habitats can be created by disturbances and also are episodically affected by them—streams are not static. Just as the influence of process drivers is not uniform across a watershed, so also is the influence of disturbances. In the case of floods, the most common and obvious of disturbances in rivers, this variability is primarily temporal and can be expressed through flood frequency, duration, seasonality (i.e., when in the year they occur), magnitude, and the predictability of occurrence (Lytle and Poff 2004).

The expression of these drivers, their interactions within a watershed and in individual streams, and their variability over time creates and maintains a dynamic mosaic of habitats. These interactions among drivers are complex and intertwined even in the absence of anthropogenic disturbances. Therefore, scientists have had difficulty separating the effects of the different drivers independently and in assessing how any species (even well studied salmon species) respond to this mosaic (Fausch et al. 2002). Adding to this complexity is the fact that habitats are species-specific; what constitutes habitat for one species may be mostly unrelated to what constitutes habitat for another aquatic species in the same river.

Thus, if riparian areas are managed with the objective of maintaining high-quality habitats for aquatic species, managers must ensure that riparian areas contribute what they can (i.e., separate from, but in combination with, watershed processes or drivers) to the long-term maintenance of the temporally variable spatial mosaic of habitat conditions. The mere presence of these habitats, however, is not sufficient: they must exist in configurations that are advantageous to aquatic species, with sufficient variety to support multiple life stages, adequate size and number to support a population, sufficiently proximate to be accessible from one to another, and yet sufficiently distal to not be subject to the same deleterious disturbances.

This, then, defines the fundamental challenge of watershed and riparian ecosystem management: given a system of great diversity and complexity, with attributes that can vary greatly and unpredictably in their influence over time, and biological communities that depend on both the diversity and interconnections between habitats, what fundamental principles should guide management to perpetuate fish and wildlife? We return to this question in the last section of this chapter.

8.4. Integrated Effects of Site-Scale Riparian Areas Management

"Connectivity" is the term most often used to describe whether organisms can move from one habitat to another (Taylor et al. 1993). It comprises two parts: structural connectivity, the physical relationships among habitat patches without regard for the behavior or organisms; and functional connectivity, the degree of movement of organisms through the landscape (Taylor et al. 2006; Kadoya 2009). The two are obviously related in all landscapes, but lotic systems impose some characteristic influences. First, there is a strong (though not universal) downstream bias on functional connectivity, insofar as most organisms and all inert constituents move in the direction of water flow. Second, impediments to flow can impose partial barriers to both structural and functional connectivity, but to very different degrees. For example, a culvert may allow most or all discharges to pass unimpeded but nevertheless present a barrier to migrating salmon (e.g., Davis and Davis 2011).

Concern over longitudinal barriers along a channel network typically focuses on the effects of instream physical barriers, particularly dams, culverts, and road crossings, rather than the consequences of longitudinal disturbances to riparian areas more broadly. Instream connectivity and riparian connectivity are related, however, even if they are not direct surrogates. Some types of discontinuity (e.g., a road running perpendicular to the stream) will affect the longitudinal connectivity (both structural and functional) of both; but others (e.g., an impassible migration barrier in the stream) will affect the movement of aquatic and terrestrial organisms quite differently. Most importantly from the perspective of riparian management, alterations that are limited to riparian areas can nonetheless affect instream (functional) connectivity. For example, loss of shading from riparian area disturbance (Chapter 4) can create unfavorable instream temperatures that impact the movement or outright survival of migrating salmonids (e.g., Thorstad et al. 2008; Martins et al. 2012). Other direct and indirect actions, such as channelization, removal of large wood, or bank failures from vegetation loss can affect connectivity for migratory fish and among other aquatic species, and alter the flow of material and nutrients through the stream network.

These impacts have been explored in a variety of studies at multiple scales. Flitcroft et al. (2012:288) reported that "…[spatial] network variables perform better at explaining juvenile Coho Salmon *Oncorhynchus kisutch* density than instream habitat variables. Moreover, analysis of network distances among seasonal habitats indicates that juvenile Coho Salmon density may be higher where the distance between critical seasonal habitats is short." Perkin and Gido (2012:2183) found lower species richness in fish communities in stream networks fragmented by road crossings across multiple spatial scales. "Our findings support network connectivity as a mediator of ecological processes occurring within complex dendritic ecosystems and promote the need for improved connectivity to enhance conservation of metacommunity dynamics and biodiversity in dendritic ecological networks." Similarly, Favaro et al. (2014:1815) interpreted their findings on the influence of culverts in a channel network to demonstrate that "…fragmentation of habitats and populations acted on a whole stream scale rather than being restricted to within-stream differences that related to position."

These findings speak to the importance of recognizing the channel as part of the riparian ecosystem, that is, they form an interconnected system, both laterally and longitudinally. The degree to which seemingly isolated discontinuities affect either of these elements directly can ultimately impact the entire ecological network. Prescriptive management is challenging, however, because the ecological functions of any given site or habitat patch along a channel network depend on its context and interconnection with other such sites. Furthermore, not all habitats have the same value for all species:

> *The relationship between species richness and connectivity is therefore determined by complex relationships among several interacting variables. In addition, species richness maxima for different faunal and floral elements occur at different positions along the connectivity gradient (Tockner et al. 1998). Fish diversity, for example, may peak in highly connected habitats, whereas amphibian diversity tends to be highest in habitats with low connectivity. Other groups attain maximum species richness between these two extremes. The resulting pattern is a series of overlapping species diversity peaks along the connectivity gradient, suggesting that habitats collectively traversing a broad range of connectivity will optimise community diversity in riverine landscapes. [From Ward et al. 2002:535.]*

Although such studies emphasize the importance of connectivity along the channel corridor, they offer little guidance for managing these barriers and discontinuities beyond *less is better.* The impacts of road crossings, in particular, have been evaluated in several studies in Pacific Northwest lowland streams, comparing measures of biological conditions with the frequency of crossings that describe a negative, monotonic relationship (May et al. 1997; McBride and Booth 2005). Unfortunately, these studies do not suggest a consistent, discrete threshold of impacts that might serve as a management objective. For example, the study of McBride and Booth (2005) showed correlations between biological health (as characterized by a multi-metric index of benthic macroinvertebrates) and road crossings with more than three crossings per km, whereas the biological differences reported by Perkin and Gido (2012) occurred on fragmented streams with as few as one road crossing over a stream segment length of one to many kilometers.

Chum Salmon Oncorhynchus keta *need connected habitat to reach spawning grounds/Alex Biswas, WDFW*

The overarching message from these studies is that any given implementation of site-scale riparian area management does not occur in a vacuum: "protecting a site" has ecological meaning only within the context of the entire channel network. Although existing computational tools can characterize the degree of fragmentation from discontinuities or the spatial coherence of a riparian corridor of varying width, the existing literature suggests that we lack clear deterministic linkages between alternative riparian management strategies and the resulting response of instream organisms, particularly the identification of thresholds of disturbance that impact fish mobility. However, any management action that results in frequent interruptions to riparian areas, or riparian corridors where blockage of terrestrial and aquatic organisms is not fully mitigated, is sure to have deleterious impacts to aquatic and terrestrial system health.

8.5. Managing Riparian Areas From a Watershed Perspective

The perspective of riparian area management developed in this chapter has been intentionally expanded, in both space and time, beyond that normally applied (and most feasibly implemented) in the Pacific Northwest. It relies heavily on the active research in the field of landscape ecology, which is organized around a few themes of particular relevance to riparian area management: 1) patches differ in quality, 2) patch boundaries affect flows, 3) patch context matters, 4) connectivity is critical, 5) organisms are important, and 6) scale is important (Wiens 2002:501). We will explore each of these in turn with an eye towards how management of riparian areas might be improved with respect to perpetuating fish and wildlife.

8.5.1. Patches Differ in Quality

Not every location along a stream is equally influential on the physical processes or ecological functions of the riparian-aquatic ecosystem. This generalization is complicated, however, by the fact that different perspectives may not assign the same level of "influence" to every location. For example, bedrock-bound channels have very limited capacity for response (Montgomery and Buffington 1997), despite the potential for highly variable, episodic inputs of water and sediment. Thus, from a geomorphological perspective they might be considered less sensitive and so "less critical" for riparian area protection. However, such channels commonly occupy headwater, high-elevation positions in their watersheds that host potentially critical habitat(s) for a variety of non-fish species. A general treatment of the topic such as this cannot resolve the multiplicity of such complex phenomena across the diversity of landscapes in the Pacific Northwest, but this brief discussion suggests the importance of recognizing the range of riparian functions that must be served, and the improbability that they are all of equivalent importance throughout a channel network.

8.5.2. Patch Boundaries Affect Flows

In general, riparian regulations rarely consider the nature of the transition from riparian to upland (i.e., from regulated to unregulated) areas. This theme provides a reminder that the character of those transitions can affect each of the patches through the flux of material, energy, and organisms across their shared boundary. So, for example, when what was once a part of a forest interior becomes the exposed edge of a buffer strip, adjacent to recently cleared uplands, windthrow can greatly increase (and so compromise the riparian management area itself). Direct adjacency of a riparian area to urban development can invite substantial human intrusion (e.g., invasive species, excess nutrients, pesticides) into the riparian area, reaching all the way to the nominally protected stream because attention is not given to the relationship of upland development to the riparian area, nor to the "porosity" of the boundary between them.

8.5.3. Patch Context Matters

Riparian areas within separate watersheds containing different land uses (e.g., forestry, agricultural, urban) provide very different ecological functions depending on the adjacent land uses. The impacts to a stream from watershed urbanization, for example, span the range from pervasive hydrologic changes (for which buffers are typically irrelevant) to pesticide-laden run-off (for which the pollutant removal that a buffer provides may be somewhat to highly effective). In a forestry dominated watershed, primary concerns may be the potential for unnaturally high delivery of sediment from increased rates of mass wasting; or potential losses of streambank integrity, habitat complexity, shading, large wood, and allochthonous inputs of organic matter.

According to Hansen et al. (2015:54):

> *Our review demonstrated that greater widths were required to achieve objectives when adjacent land use intensity was high, or when the objective of management was improving terrestrial biodiversity (particularly fauna). This becomes problematic when intense land use practices occur on small properties, reducing the amount of riparian land that can economically be protected or targeted for management...For example, if we apply the evidence summarized here to a streamside property used for dairying in the lower Hunter River, New South Wales, a riparian zone width of 40 m may achieve ≥75% reduction of nitrogen inputs to the river and reduce streambank erosion, contributing to improved downstream water quality. However, the same investment in riparian set-aside in the upper reaches of the Hunter catchment may provide additional improvements to stream nutrient processing (Lowe and Likens 2005), aquatic biodiversity (Chessman et al. 1997) and bird diversity (Bennett et al. 2014).*

8.5.4. Connectivity is Critical

Connectivity in riparian areas occurs not only parallel to the stream (previous section), but also orthogonally to the channel in a lateral dimension – from the stream through the riparian area into uplands—and the vertical dimension in the hyporheic zone. Typical riparian area regulations are generally successful at maintaining connectivity in the longitudinal dimension at some minimum width, although the consequences of limited interruptions to longitudinal connectivity (e.g., bridges, utility crossings) are commonly dismissed, a consequence of viewing riparian areas only from the perspective of lateral width rather than a recognition of the importance of stream network connectivity.

Longitudinal connectivity is especially important for the movement of sediment, large wood, and other organic matter. Fox (2003), for instance, found that the condition of riparian areas was a reasonable predictor for the quantity of large wood key pieces in adjacent stream channels; however, upstream basin characteristics and processes were likely responsible for most of the large wood in larger channels. Therefore, land managers should not base site-level objectives for in-stream large wood on the condition of adjacent riparian areas alone, but should also manage riparian areas at the watershed scale to provide adequate in-stream large wood throughout a watershed's stream network.

Issaquah Creek is a tributary to Lake Sammamish in King County/ Derek Booth

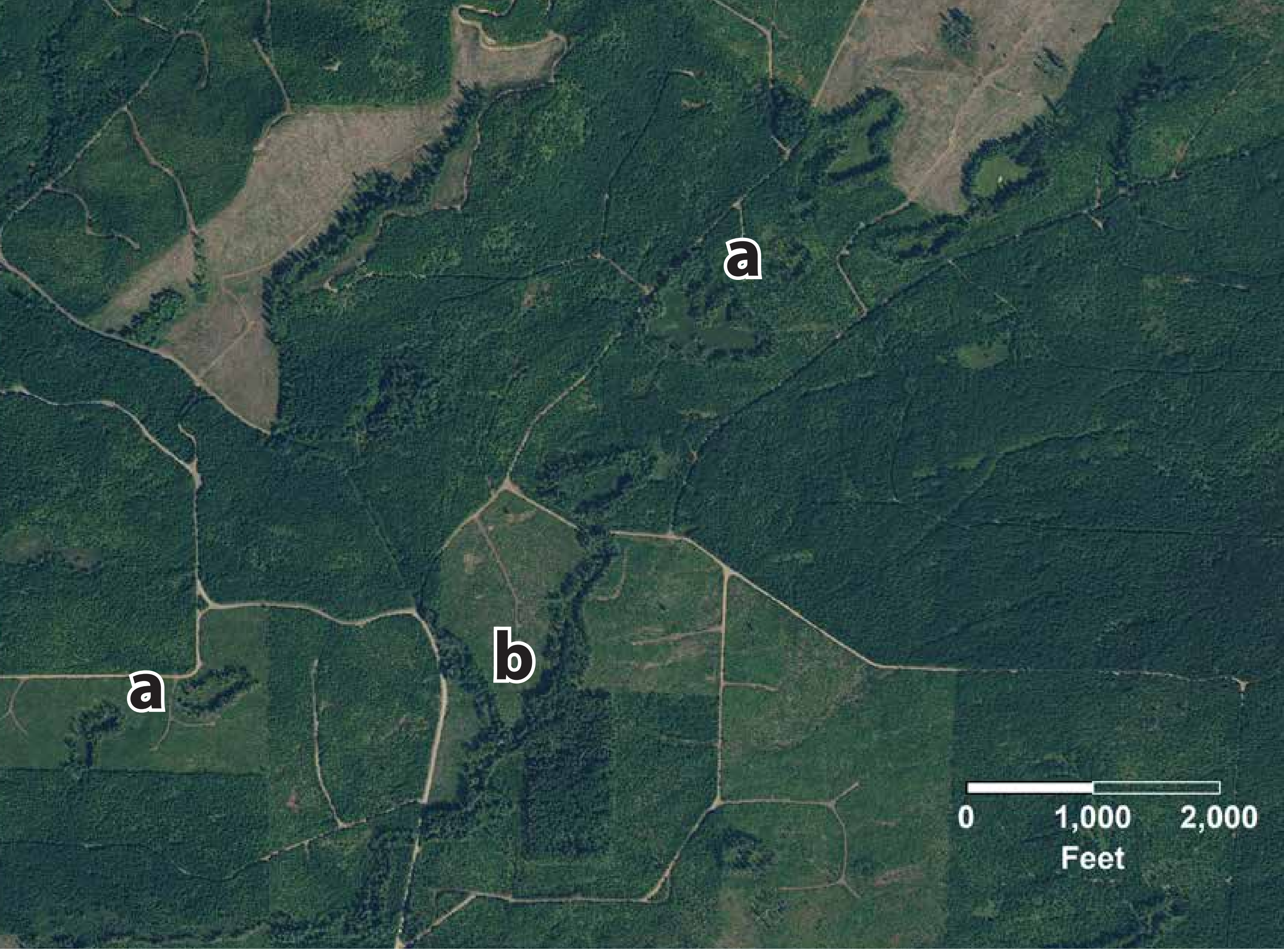


Figure 8.8. View of isolated wetlands (a) and riparian management zones (b) in a commercially logged portion of southwest Washington, providing some degree of local, structural connectivity within each buffer but no functional connectivity between them. Forested buffers in this image are approximately 50 feet (15 meters) wide.

Lateral connectivity, particularly between the riparian area and the adjacent uplands, is a more complex relationship, because improving this connectivity simultaneously enhances ecosystem protection but can also compromise the protection of a stream from upland activities. Continuity of a protected riparian area with uplands that are protected as well provides a range of key habitats for mobile species that depend on both upland and riparian environments for different needs or at different life stages. Separated patches with relatively porous boundaries may achieve only a modest fraction of these benefits, however, and truly isolated riparian–upland patches may be no better than an absence of such habitats altogether (at least for many terrestrial species of concern) (Figure 8.8). Porous boundaries also run the risk of compromising some of the primary goals of a riparian management area, and so the nature of connectivity does not universally result in beneficial outcomes (see 8.5.2 above).

Vertical connectivity occurs at the ecotone between stream water and deeper groundwater. This ecotone, known as the hyporheic zone, is comprised of coarse sediments, and the vertical connectivity occurring there is more commonly referred to as hyporheic exchange (Boultan et al. 1998). Hyporheic exchange includes water, dissolved solutes, fine sediments, fine organic matter, and small organisms. The size of the hyporheic zone is enlarged by structures such as large wood and boulders that store sediment. Hyporheic exchange is enhanced by structures such as large wood or boulders that redirect flow into the streambed and by undulations in channel morphology (e.g., meanders, pool-riffle beds) that intercept the flow of water.

In summary, the importance of connectivity is broadly recognized:

> *Restoration of important ecological processes often implies improving connectivity of the stream. For example, longitudinal and lateral connectivity can be enhanced by restoring fluvial dynamics on flood-suppressed rivers and by increasing water availability in rivers subject to water diversion or withdrawal, thereby increasing habitat and species diversity. Restoring links between surface and ground water flow enhances vertical connectivity and communities associated with the hyporheic zone. [From Jansson et al. 2007:589.]*

However, tangible recommendations for how best to manage for this outcome in typical regulatory settings are more elusive.

8.5.5. Organisms are Important

Although not strictly an issue of spatial or temporal relationships between landscape elements, the recognition of organisms as key elements of riverine composition, structure, and functions is a valuable contribution of landscape ecology to watershed management. Some of these influences have been long-appreciated: the importance of large wood to stream structure and salmonid habitat, the role of root strength in mediating bank erosion, and indeed the entire construct of conserving riparian areas of native vegetation to maintain ecosystem health. However, this perspective has broadened our appreciation of the multiple ways in which biota interact with the physical environment and each other to support the full range of watershed processes and thus the systems that depend on them (Beavers, bears, and marine nutrients, etc.). Management of riparian areas thus must include not only some awareness of the watershed in which that riparian area exists but also the biota that inhabit this environment (and, in some cases, move beyond its boundaries), and the ways in which those biota support the integrity of the aquatic ecosystem (Robinson et al. 2002).

Black Bear Ursus Americanus *feeding on grass/Scott Fitkin, WDFW*

8.5.6. Scale is Important

The goals and application of riparian management actions are generally limited in both space (i.e., the site) and time (i.e., the present). In a system as interconnected and disturbance-prone as a stream channel network, however, the site-scale management of a riparian area is *necessary*, but not *sufficient*, to ensure the desired composition, structure, and functions of riparian and aquatic habitats. Of all locations on a natural or human-dominated landscape, river channels are probably the least stable in time, experiencing (and depending on) both gradual changes and episodic disturbance that structure their habitats and the organisms that utilize them. They also are subject to a spatial hierarchy of influences, not only those of the immediately adjacent riparian area itself (such as shade) but also those of the upstream watershed (delivering water and sediment to the channel) and the landscape even beyond its drainage boundaries. "Conditions far from stream banks affect the distribution of key instream habitat characteristics. Amount of instream wood, percentage of gravel, and pool frequency, which are essential to healthy salmon habitat, are particularly sensitive to land use" (O'Callaghan 2012:2). At even larger scales, the mobility of large terrestrial animals, the ability of salmon to return to headwater streams from the ocean, and the episodic disturbance from wildfire can exert equivalent (or greater) influences.

8.6. Conclusions

The perspective provided by landscape ecology emphasizes the temporally and spatially varying nature of habitats, and it suggests that achieving genuine habitat protection (and thus species conservation) is crippled by management that is limited to only narrow riparian corridors. To perpetuate fish and wildlife in the long term, three principles need to become integrated into the current paradigm for protecting aquatic ecosystems:

1. More important than simply protecting habitats is protecting the processes that create and sustain habitats. For this strategy to be successful, however, space in which that habitat can be expressed must also be protected. So, for example, instream and riparian area protection is indeed necessary, but it is not sufficient—the watershed and local-scale processes that create habitats within these areas must also be maintained. Current paradigms that focus on site-level management of riparian areas will not successfully support the dynamic processes that maintain diverse, productive fluvial ecosystems.

2. Habitats have not only spatial but also temporal dimensions to their creation and maintenance, and they cannot retain their functions if they remain static. The historical frequencies, durations, and magnitudes of natural disturbances (flood and fire being the most common) need to be better understood and then emulated to the greatest extent that surrounding land uses can tolerate.

3. Spatial connectivity between habitats is critical, particularly for species (such as salmon) with a variety of habitat requirements at different life stages. Isolated key habitats for mobile species are scant improvement over no habitat at all.

Integration of these principles into the management of aquatic systems will be challenging under existing regulatory frameworks. However, even an awareness of their relevance to sustaining healthy ecosystems may reveal opportunities for their application in regulatory and non-regulatory settings alike. The following key points are offered in support of such opportunities:

Decaying salmon carcass/Ned Pitmann, WDFW

8.6.1. Manage the Riverine Landscape

The riverine landscape is the proper construct for understanding and managing aquatic systems. Although riparian areas are disproportionately important for fish and wildlife relative to their area on a landscape, they are not exclusively important. Although riparian areas affect many critical instream conditions, including temperature, nutrient input, bank stability, and large wood, while also filtering sediment and other pollutants from adjacent upland areas, the watershed outside of the riparian area is no less influential on the ultimate state of instream conditions, particularly with respect to its control of hydrology and sediment delivery. From even beyond a watershed's boundaries, influences on the stream can be carried by wind, mobile biota, and wildfire.

8.6.2. Manage at Multiple Scales

Effective and efficient conservation of fish and wildlife habitats requires management at multiple spatial and temporal scales. Just as the influences on streams arise from within and beyond the riparian area, management must find vehicles to embrace these multiple spatial scales as well. Treatments must also embrace the ever-changing temporal patterns of stability and disturbance, which not only result in constantly shifting physical habitats but also create the ecological template necessary for healthy, resilient biological communities (Ryan and Calhoun 2010). This will require approaches that allow sufficient space and for the expression of key habitat-forming processes (Beechie et al. 2010) rather than simply the creation or protection of static habitat features.

8.6.3. Connect Individual Actions

Particularly in rivers, longitudinal connectivity of individual locations is a defining feature of these systems and their biota, and so maintaining this connectivity is an essential consideration of aquatic ecosystem protection. We increase our chances of maintaining connectivity with coordinated actions across all riparian sites, and even more so by considering the entire riverine landscape as the focus of conservation efforts. However, connectivity is only as strong as its weakest link, and so even a single ill-executed action can preclude achieving this goal altogether.

8.6.4. Engage the Challenge of Implementing Effective Management

Adhering to these key points for achieving genuine riparian ecosystem protection, particularly in the face of continued human impacts, may not be presently attainable. Even if specific beneficial actions are neither everywhere known nor universally feasible to implement, our current understanding can support a positive trajectory. "Landscape conditions far from the river's edge may have strong impacts on instream conditions. Understanding the effects of natural processes and human activities across entire drainage basins is key to researching, monitoring, and restoring aquatic resources." (O'Callaghan 2012:5). Progress will require not only an appreciation of the most effective riparian management strategies but also a broadened perspective on the scope of necessary actions, actions that must extend far beyond the river itself and its adjacent riparian ecosystem.

8.7. Literature Cited[1]

Agouridis, C.T., S.R. Workman, R.C. Warner, and G.D. Jennings. 2005. Livestock grazing management impacts on stream water quality: a review. Journal of the American Water Resources Association 41:591-606. (i)

Allan, J.D. 2004. Landscapes and riverscapes: the influence of land use on stream ecosystems. Annual Review of Ecology, Evolution, and Systematics 35:257-284. (i)

Andréassian, V. 2004. Waters and forests: from historical controversy to scientific debate. Journal of Hydrology 291:1-27. (i)

Anlauf, K.J., D.W. Jensen, K.M. Burnett, E.A. Steel, K. Christiansen, J.C. Firman, B.E. Feist, and D.P. Larsen. 2011. Explaining spatial variability in stream habitats using both natural and management-influenced landscape predictors. Aquatic Conservation: Marine and Freshwater Ecosystems 21:704-714. (i)

Arkle, R.S., D.S. Pilliod, and K. Strickler. 2010. Fire, flow and dynamic equilibrium in stream macroinvertebrate communities. Freshwater Biology 55:299-314. (i)

Beechie, T.J., D.A. Sear, J.D. Olden, G.R. Pess, J.M. Buffington, H. Moir, P. Roni, and M.M. Pollock. 2010. Process-based principles for restoring river ecosystems. BioScience 60:209-222. (i)

Belsky, A.J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54:419-431. (i)

Bennett A.F., D.G. Nimmo, and J.Q. Radford. 2014. Riparian vegetation has disproportionate benefits for landscape-scale conservation of woodland birds in highly modified environments. Journal of Applied Ecology 51:514-523. (i)

Booth, D.B., J.R. Karr, S. Schauman, C.P. Konrad, S.A. Morley, M.G. Larson, and S.J. Burges. 2004. Reviving urban streams: land use, hydrology, biology, and human behavior. Journal of the American Water Resources Association 40:1351-1364. (i)

Boulton, A.J., S. Findlay, P. Marmonier, E.H. Stanley, and H.M. Valett. 1998. The functional significance of the hyporheic zone in streams and rivers. Annual Review of Ecology and Systematics 29:59-81. (i)

Brown, L.R., T.F. Cuffney, J.F. Coles, F. Fitzpatrick, G. McMahon, J. Steuer, A.H. Bell, and J.T. May. 2009. Urban streams across the USA: lessons learned from studies in 9 metropolitan areas. Journal of the North American Benthological Society 28:1051-1069. (i)

Chessman B.C., J.E. Growns, and A.R. Kotlash. 1997. Objective derivation of macroinvertebrate family sensitivity grade numbers for the SIGNAL biotic index: application to the Hunter River system, New South Wales. Marine and Freshwater Research 48:159-172. (i)

Coffman, G.C., R.F. Ambrose, and P.W. Rundel. 2010. Wildfire promotes dominance of invasive Giant Reed (*Arundo donax*) in riparian ecosystems. Biological Invasions 12:2723-2734. (i)

Davis, J.C., and G.A. Davis. 2011. The influence of stream-crossing structures on the distribution of rearing juvenile Pacific salmon. Journal of the North American Benthological Society 30:1117-1128. (i)

Fausch, K.D., C.E. Torgersen, C.V. Baxter, and L.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52:483-498. (i)

Favaro, C., J.W. Moore, J.D. Reynolds, and M.P. Beakes. 2014. Potential loss and rehabilitation of stream longitudinal connectivity: fish populations in urban streams with culverts. Canadian Journal of Fisheries and Aquatic Sciences 71:1805-1816. (i)

[1] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Flitcroft, R.L., K.M. Burnett, G.H. Reeves, and L.M. Ganio. 2012. Do network relationships matter? Comparing network and instream habitat variables to explain densities of juvenile Coho Salmon (*Oncorhynchus kisutch*) in mid-coastal Oregon, USA. Aquatic Conservation: Marine and Freshwater Ecosystems 22:288-302. (i)

Fox, M.J. 2003. Spatial organization, position, and source characteristics of large woody debris in natural systems. Doctoral dissertation. University of Washington, Seattle. (i)

Frissell, C.A., W.J. Liss, C.E. Warren, and H.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199-214. (i)

Hansen, B.D., P. Reich, T.R. Cavagnaro, and P.S. Lake. 2015. Challenges in applying scientific evidence to width recommendations for riparian management in agricultural Australia. Ecological Management and Restoration 16:50-57. (i)

Jansson, R., C. Nilsson, and B. Malmqvist. 2007. Restoring freshwater ecosystems in riverine landscapes: the roles of connectivity and recovery processes. Freshwater Biology 52:589-596. (i)

Jellyman, P.G., D.J. Booker, and A.R. Mcintosh. 2013. Quantifying the direct and indirect effects of flow-related disturbance on stream fish assemblages. Freshwater Biology 58:2614-2631. (i)

Kadoya, T. 2009. Assessing functional connectivity using empirical data. Population Ecology 51:5-15. (i)

Keeley J.E., G.H. Aplet, N.L. Christensen, S.G. Conard, E.A. Johnson, P.N. Omi, D.L. Peterson, and T.W. Swetnam. 2009. Ecological foundations for fire management in North American forest and shrubland ecosystems. PNW-GTR-779. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Lake, P.S. 2000. Disturbance, patchiness, and diversity in streams. Journal of the North American Benthological Society 19:573-592. (i)

Lowe, W.H., and G.E. Likens. 2005. Moving headwater streams to the head of the class. BioScience 55:196–197. (i)

Luce, C., P. Morgan, K. Dwire, D. Isaak, Z. Holden, and B. Rieman. 2012. Climate change, forests, fire, water, and fish: building resilient landscapes, streams, and managers. RMRS-GTR-290. U.S. Forest Service, Rocky Mountain Research Station, Fort Collins, Colorado. (viii)

Lytle, D.A., and N.L. Poff. 2004. Adaptation to natural flow regimes. Trends in Ecology and Evolution 19:94-100. (i)

Malanson, G.P. 1995. Riparian landscapes. Cambridge University Press, Cambridge, United Kingdom. (i)

Martins, E.G., S.G. Hinch, D.A. Patterson, M.J. Hague, S.J. Cooke, K.M. Miller, D. Robichaud, K.K. English, and A.P. Farrell. 2012. High river temperature reduces survival of Sockeye Salmon (*Oncorhynchus nerka*) approaching spawning grounds and exacerbates female mortality. Canadian Journal of Fisheries and Aquatic Sciences 69:330-342. (i)

May, C.L. 2002. Debris flows through different forest age classes in the central Oregon Coast Range. Journal of the American Water Resources Association 38:1097-1113. (i)

May, C.W., R.R. Horner, J.R. Karr, B.W. Mar, and E.B. Welch. 1997. Effects of urbanization on small streams in the Puget Sound Lowland Ecoregion. Watershed Protection Techniques 2:483-494. (i)

McBride, M., and D.B. Booth. 2005. Urban impacts on physical stream conditions: effects of spatial scale, connectivity, and longitudinal trends. Journal of the American Water Resources Association 41:565–580. (i)

Montgomery, D.R. 1999. Process domains and the river continuum. Journal of the American Water Resources Association 35:397-410. (i)

Montgomery, D.R., and J.M. Buffington. 1997. Channel-reach morphology in mountain drainage basins Bulletin of the Geological Society of America 109:596-611. (i)

Montgomery, D.R., J.M. Buffington, N.P. Peterson, D. Schuett-Hames, and T.P. Quinn. 1996. Stream-bed scour, egg burial depths, and the influence of salmonid spawning on bed surface mobility and embryo survival. Canadian Journal of Fisheries and Aquatic Sciences 53:1061-1070. (i)

Moore, J.W. 2006. Animal ecosystem engineers in streams. BioScience 56:237-246. (i)

Morley, S.A., and J.R. Karr. 2002. Assessing and restoring the health of urban streams in the Puget Sound Basin. Conservation Biology 16:1498-1509. (i)

O'Callaghan, J. 2012. Thinking big: linking rivers to landscapes. Science Findings, Issue 139. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Osborne, L.L., and D.A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243-258. (i)

Pausas, J.G., and J.E. Keeley. 2014. Abrupt climate-independent fire regime changes. Ecosystems 17:1109-1120. (i)

Perkin, J.S., and K.B. Gido. 2012. Fragmentation alters stream fish community structure in dendritic ecological networks. Ecological Applications 22:2176-2187. (i)

Pollock, M.M., M. Heim, and D. Werner. 2003. Hydrologic and geomorphic effects of Beaver dams and their influence on fishes. American Fisheries Society Symposium 37:213-233. (i)

Raymond, K.L., and B. Vondracek. 2011. Relationships among rotational and conventional grazing systems, stream channels, and macroinvertebrates. Hydrobiologia 669:105-117. (i)

Reid, L.M., N.J. Dewey, T.E. Lisle, and S. Hilton. 2010. The incidence and role of gullies after logging in a coastal redwood forest. Geomorphology 117:155-169. (i)

Richardson, J.S., R.E. Bilby, and C.A. Bondar. 2005. Organic matter dynamics in small streams of the Pacific Northwest. Journal of the American Water Resources Association. 41:921-934. (i)

Robinson, C.T., K. Tockner, and J.V. Ward. 2002. The fauna of dynamic riverine landscapes. Freshwater Biology 47:661-677. (i)

Roper, B.B., B. Jarvis, and J.L. Kershner. 2007. The role of natural vegetative disturbance in determining stream reach characteristics in central Idaho and western Montana. Northwest Science 81:224-238. (i)

Ryan, D.F., and J.M. Calhoun. 2010. Riparian adaptive management symposium: a conversation between scientists and management. PNW-GTR-830. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Stanley, S., S. Grigsby, D.B. Booth, D. Hartley, R. Horner, T. Hruby, J. Thomas, P. Bissonnette, R. Fuerstenberg, J. Lee, P. Olson, and G. Wilhere. 2011. Puget Sound characterization. volume 1: the water resources assessments (water flow and water quality). Publication #11-06-016. Washington State Department of Ecology, Olympia. Available: https://fortress.wa.gov/ecy/publications/publications/1106016.pdf. (January 2019). (viii)

Talbot L.M. 1996. The linkage between ecology and conservation policy. Pages 368-378 *in* S.T.A. Pickett, R.S. Ostfeld, M. Shachak, and G.E. Likens, editors. The ecological basis of conservation. Chapman & Hall, New York. (i)

Taylor, P.D., L. Fahrig, K. Henein, and G. Merriam. 1993. Connectivity is a vital element of landscape structure. Oikos 68:571-573. (i)

Taylor P.D., L. Fahrig, and K.A. With. 2006. Landscape connectivity: a return to the basics. Pages 29–43 *in* K. Crooks and M.A. Sanjayan, editors. Connectivity conservation. Cambridge University Press, Cambridge, United Kingdom. (i)

Thorstad, E.B., F. Økland, K. Aarestrup, and T.G. Heggberget. 2008. Factors affecting the within-river spawning migration of Atlantic Salmon, with emphasis on human impacts. Reviews in Fish Biology and Fisheries 18:345-371. (i)

Tockner, K., F. Schiemer, and J.V. Ward. 1998. Conservation by restoration: the management concept for a river-floodplain system on the Danube River in Austria. Aquatic Conservation: Marine and Freshwater Ecosystems 8:71-86. (i)

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, and C.E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137. (i)

Vietz, G.J., M.J. Sammonds, C.J. Walsh, T.D. Fletcher, I.D. Rutherfurd, and M.J. Stewardson. 2014. Ecologically relevant geomorphic attributes of streams are impaired by even low levels of watershed effective imperviousness. Geomorphology 206:67-78. (i)

Wallbrink, P.J., B.P. Roddy, and J.M. Olley. 2002. A tracer budget quantifying soil redistribution on hillslopes after forest harvesting. Catena 47:179-201. (i)

Walsh, C.J., A.H. Roy, J.W. Feminella, P.D. Cottingham, P.M. Groffman, and R.P. Morgan II. 2005. The urban stream syndrome: current knowledge and the search for a cure. Journal of the North American Benthological Society 24:706-723. (i)

Wang, L., J. Lyons, P. Kanehl, and R. Bannerman. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environmental Management 28:255-266. (i)

Ward, J.V., K. Tockner, D.B. Arscott, and C. Claret. 2002. Riverine landscape diversity. Freshwater Biology 47:517-539. (i)

Wiens, J.A. 2002. Riverine landscapes: taking landscape ecology into the water. Freshwater Biology 47:501-515. (i)

Chapter 9. Science Synthesis to Management Implications

By: Timothy Quinn, Kirk L. Krueger, and George F. Wilhere

9.1. Introduction

This chapter summarizes important scientific findings from chapters 1 through 8 and provides a brief discussion of scientific themes that can inform thoughtful approaches to protecting or restoring riparian ecosystems and associated aquatic habitats. The current state of the science, as reviewed in chapters 1 through 8, clearly demonstrates the importance of an intact riparian ecosystem to the proper functioning of aquatic habitats. It also introduces new studies on conservation of watershed-scale processes that directly affect aquatic habitats and that help maximize the ecological benefits of riparian ecosystems.

Riparian ecosystems are a priority habitat because their composition, structure, and functions dramatically affect a multitude of fish, amphibian, reptile, bird, mammal, and invertebrate species (Cummins 1974; Richardson et al. 2005). Although riparian ecosystems are a small portion of the landscape, approximately 85% of Washington's wildlife species use them (Thomas et al. 1979). Protecting or restoring high function to this relatively small portion of the landscape can disproportionally benefit many species and other important ecosystem goods and services (e.g., clean water, fisheries, and flood control) (NRC 2002).

Our definition of riparian ecosystem is integral to the development of management recommendations. Riparian ecosystems are transitional areas (ecotones) between terrestrial and aquatic ecosystems and are distinguished by gradients in biophysical conditions, ecological processes, and biota (NRC 2002). Riparian ecosystems include areas through which surface and subsurface hydrology connect waterbodies with uplands. They include those portions of terrestrial area that significantly influence exchanges of energy and matter with aquatic ecosystems (i.e., a zone of influence). Following Naiman and Bilby (1998), we include active floodplains and the terraces and adjacent uplands that contribute organic matter to active channels or active floodplains as part of the riparian ecosystem.

9.2. Summary of Chapters 2 Through 8

Chapters 2 through 8 discussed five key riparian ecosystem functions including bank stability, stream shading, wood recruitment, litter fall, and pollutant removal, as well as other topics related to the riparian ecosystems including stream channel morphology, nutrient dynamics, dryland riparian areas, and watershed-scale processes. Here we summarize the main points of each chapter.

9.2.1. Stream Channel Morphology (Chapter 2)

The stream morphology chapter explained how the composition, structure, and functions of riparian ecosystems and their associated aquatic habitats depend on dynamic interactions of water, sediment, wood, and riparian vegetation. Important points from this chapter are:

- Erosion, transport, and deposition of sediment are part of a hierarchy of disturbance processes that continually create, maintain, destroy, and recreate the variety of aquatic habitats.

- Geomorphic processes related to the input of sediments to streams (from local stream bank erosion to mass wasting events) are integral to a watershed's natural disturbance regime and necessary for long-term ecological sustainability. For example, sediments delivered to streams from hillslope mass wasting events are often the predominant source of gravel, which is essential for salmon spawning habitat.

- Bank stability is one of five key ecological functions of riparian areas. Vegetation has mechanical and hydrologic effects on bank stability, and these effects can be stabilizing or destabilizing. However, in general, the net effect of vegetation is stabilizing.

- Simple models suggest that in an old-growth riparian forest, the full contribution of root strength to streambank stability is provided by trees within approximately 33 ft (10 m) of the streambank.

- Over a period of decades to centuries, a river or stream will travel across the entirety of its channel migration zone. Because the riparian ecosystem is defined by proximity to water, as the active channel moves back and forth across the channel migration zone, the riparian ecosystem moves with it.

9.2.2. Wood (Chapter 3)

Chapter 3 focused on the role of wood in aquatic ecosystems, and the recruitment of large wood to aquatic ecosystems. Wood plays a critical role in the composition, structure, and functions of riparian and aquatic ecosystems. In-stream large wood increases hydraulic complexity, i.e., creates a wider range of flow velocities, which causes pool formation, streambed scour, sediment deposition, and channel migration. The net result is a diversity of aquatic habitat types. Main points from the wood chapter include:

- The effects of wood on aquatic ecosystems in the Pacific Northwest are well-understood and ecologists are very confident about the critical role of wood in fish habitats.

- Ecologists are also confident about the role of riparian forests in supplying large wood to aquatic systems.

- Source distances for in-stream wood are a function of tree height in riparian ecosystems. Source distances are longer for riparian forests with taller trees.

- The main uncertainties for management are: the shape of the wood recruitment function, and in particular, the shape under different site-level and watershed conditions; and the potential contributions from recruitment mechanisms outside the riparian forest, such as landslides, debris flows, and extreme channel migration.

- In an unmanaged natural riparian area, 100% of a site's large wood recruitment potential comes from a distance of about 85 to 230 ft (26 to 70 m), depending on site productivity class. This does not account for additional wood recruitment through landslides, debris flows, or extreme channel migration.

9.2.3. Stream Temperature (Chapter 4)

The chapter on stream temperature presented a conceptual model for stream thermal regimes, discussed the effects of water temperature on fish, herpetofauna, and invertebrates; and reviewed the impacts of various land uses on stream thermal regimes. The chapter's key points include:

- Riparian vegetation can strongly affect stream thermal regimes (i.e., the spatial and temporal distribution of temperatures) which affects the survival and productivity of aquatic species. The importance of the thermal regime, not just summer peak temperatures, to aquatic species is increasingly acknowledged, but our ability to manage for suitable thermal regimes is limited.

- Fish and other aquatic organisms use different stream habitats, defined in part by specific water temperature ranges. However, the scientific literature is conclusive about the importance of cool stream temperatures (≤60°F [15.6°C]) for salmonid survival and productivity.

- Shade can substantially reduce the amount of direct solar radiation to the stream, which is one of the main sources of heating, especially in narrow low-order streams.

- Shade from riparian vegetation is affected by vegetation height, canopy density, and stem density. The width of a riparian area affects the amount of shading to the stream.

- A stream's thermal regime is also affected by latitude, elevation, aspect, topographic shading, channel bed shape, and water volume in addition to tributary, groundwater, and hyporheic flows.

- Structures such as large wood can reduce stream temperature by increasing pool depths, causing the development of undercut banks that provide shade, and by enhancing hyporheic exchange through redirecting surface flow to subsurface flow.

9.2.4. Pollution Removal (Chapter 5)

The pollutant removal function is unique in that it only exists in the presence of human activities that generate polluted water, and it is only necessary when nonpoint source runoff from upland activities threaten to degrade water quality. The pollution removal chapter covered the common sources, biological impacts, in situ chemistry, transport pathways, and results of removal efficacy studies for five types of pollutants—sediments, excess nutrients, metals, pesticides and other organic compounds, and pathogens. Key points from the pollution removal chapter include:

- There is overwhelming evidence in the scientific literature that riparian buffers[1] reduce nonpoint source water pollution for a variety of pollutants, and that riparian buffer width is the most important variable explaining removal efficacy.

- Research also indicates that the second most important variable in pollution removal is the vegetative composition and structure in riparian buffers.

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[1] Riparian buffer refers to a strip of land adjacent to a stream that helps protect the stream from nonpoint source pollutants originating from upslope areas.

- Riparian buffer width recommendations depend on the particular pollutant to be removed and the desired removal efficacy. Policy makers must choose the desired removal efficacy.

- Models from four separate peer-reviewed meta-analyses show that riparian buffer widths for 95% removal efficacy of sediment range from 56 to 251 ft (17 to 76 m). Meta-analyses for other pollutants exhibit a similar degree of uncertainty with regard to riparian buffer width.

- Due to uncertainty regarding riparian buffer widths needed for a desired removal efficacy, management of riparian areas for the removal of pollutants should be monitored within an adaptive management program.

9.2.5. Nutrient Dynamics (Chapter 6)

The nutrient dynamics chapter explained the many effects of riparian areas on the three primary macro-nutrients: nitrogen, phosphorus, and carbon. Riparian areas can be nutrient sources or sinks, and riparian areas mediate the movements of nutrients between uplands and surface waters. Key points from the nutrient dynamics chapter include:

- Organic matter from riparian areas is an essential source of nutrients and chemical energy for aquatic ecosystems. Nutrients enter streams via plant litter fall or via surface and subsurface water flow.

- Streams in the Pacific Coastal Ecoregion of Washington (Naiman et al. 2000) are generally nutrient-poor and well shaded. Hence, additional nutrients (e.g., salmon carcasses) may enhance aquatic ecosystem productivity. However, aquatic communities may be adversely impacted by excessive anthropogenic nutrients such as nitrogen and phosphorus.

- One study suggests that riparian area widths needed to deliver 95% of annual litter input from mature conifer forests to streams ranges from 39–54% of mean tree height depending on a site's slope and wind exposure.

- Empirical evidence suggests increasing salmon carcasses, and their marine-derived nutrients and energy, to riparian ecosystems could contribute significantly to the recovery of imperiled salmon populations.

- More research is needed on the effects of riparian litter fall on stream productivity and on the relationship between salmon harvest escapement and salmon population productivity.

9.2.6. Riparian Areas of the Columbia Plateau (Chapter 7)

The chapter on riparian areas of the Columbia Plateau covered aspects of riparian ecology that are unique to this semi-arid to arid (i.e., dryland) ecoregion. The chapter clarified the definition of riparian ecosystem in dryland ecoregions, and reviewed the history of human impacts to riparian areas in the Columbia Plateau. It also reviewed the scientific literature on the five key riparian ecosystem functions – bank stability, stream shading, wood recruitment, litter fall, and pollutant removal – and it added a function of particular importance in the Columbia Plateau – alluvial water storage. Other important points include:

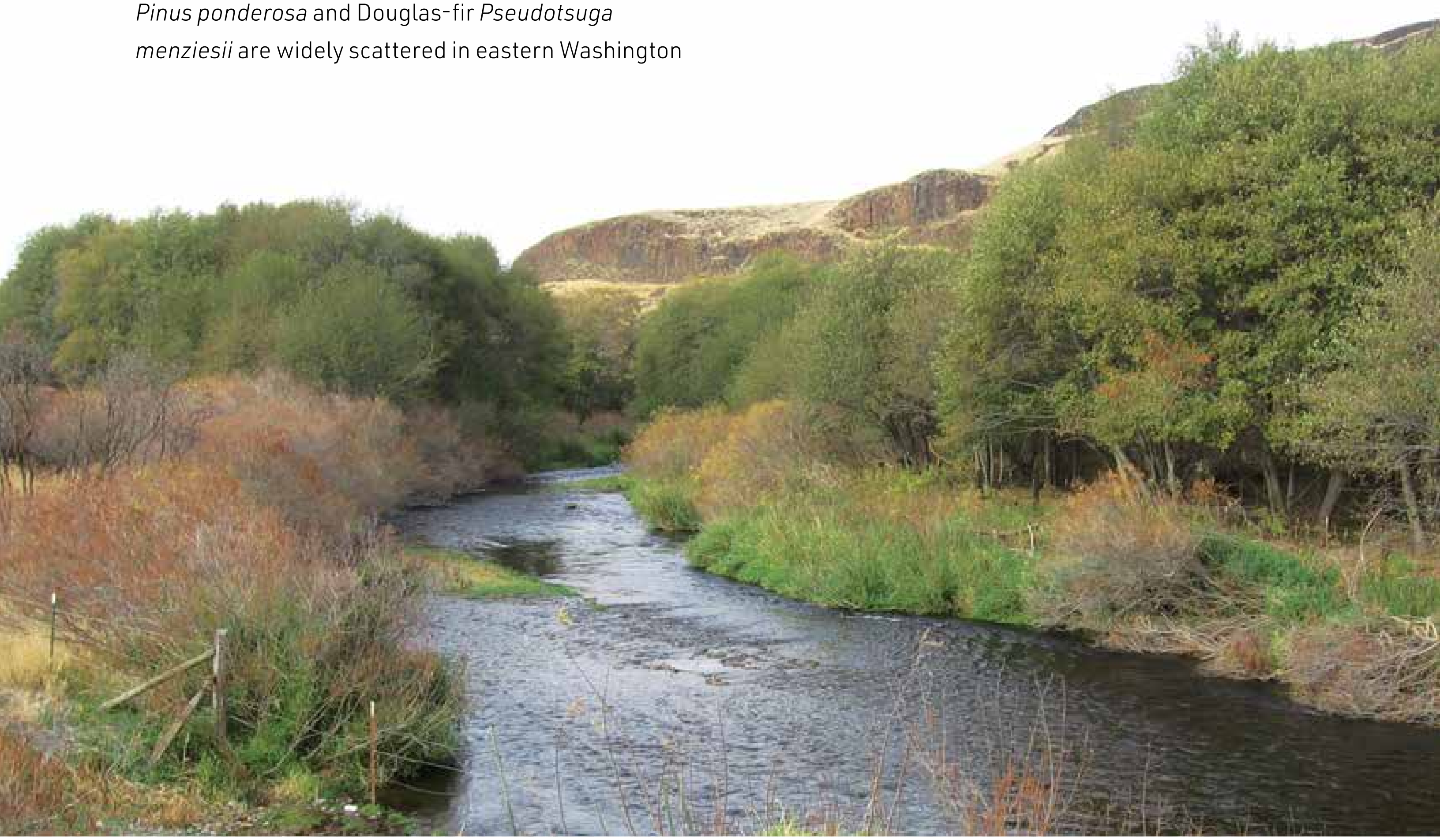
- Vegetation within riparian ecosystems of the Columbia Plateau often exhibits an abrupt demarcation between the riparian zone and the zone of influence. Phreatophytic trees and shrubs and hydrophytic herbaceous plants are confined to moist streamside areas, but the upland zone of influence may consist of sagebrush or bunchgrass communities.

- Nearly all riparian areas in the Columbia Plateau have been significantly impacted by human land use (grazing, intensive agriculture), resource exploitation (Beaver *Castor canadensis* trapping, timber harvest), water management (dams, diversions, reservoirs), or invasive plant species.

- Riparian areas in the Columbia Plateau are inhabited by a variety of deciduous woody plants, with the largest being Black Cottonwood *Populus trichocarpa* and White Alder *Alnus rhombifolia*. Pondersoa Pine *Pinus ponderosa* and Douglas-fir *Pseudotsuga menziesii* are widely scattered in eastern Washington riparian areas and were likely more common historically than at present. Compared to forested ecoregions, large woody debris play a much smaller role in fluvial processes.

- A rudimentary classification system based on overstory conveys obvious differences in stream shading provided by riparian areas: tall tree, short tree, tall shrub, shrub, grass-like, grass, and forb. The obvious implication of these vegetation types is that some stream channels in the Columbia Plateau receive little or no shade from vegetation, even historically. However, the actual historical condition of many riparian areas is unknown.

- Where the riparian zone is narrow (<100 ft [<30 m]) or the zone of influence lacks tall trees (<100 ft), the width of the riparian ecosystem is based on the pollutant removal function.



The Tucannon River flows generally northwest from headwaters in the Blue Mountains of southeastern Washington to the Snake River/Terra Hegy, WDFW

9.2.7. Watersheds (Chapter 8)

The chapter on watershed-scale processes looked beyond the site-level perspective at which riparian ecosystems are normally studied and managed. The chapter presented a conceptual framework that identified the key watershed processes and their primary influences on riparian ecosystems. The key points from the watersheds chapter are:

- Riparian ecosystems occupy perhaps the most dynamic and disturbance-prone area in a watershed. Nevertheless, the relative abundances of various aquatic habitat types in unmanaged, natural watersheds tends to remain constant over long periods (decades to centuries). In other words, at the watershed scale, habitat types appear to be in dynamic equilibrium.

- Riparian areas and surrounding watersheds are complex and dynamic systems comprised of many interacting components. These interactions across the watershed and through time create a mosaic of stream conditions necessary for self-sustaining populations of fish and other aquatic organisms.

- Some animal species have major impacts on the composition and structure of stream networks. Salmon, for example, restructure channel bed substrates during spawning and Beaver restructure the morphology and hydrology of floodplains. These “ecosystem engineers” are vital components of riparian ecosystems.

- Longitudinal, lateral, and vertical connectivity are critical for maintaining or restoring riparian ecological composition, structure, and functions. Studies have shown the importance of connectivity along the riparian corridor, but are often lacking specific guidance for management beyond “more is better.”

- The composition, structure, and functions of riparian areas are affected by surrounding land uses (e.g., forestry, agriculture, residential).

- The effectiveness of riparian ecosystems to protect aquatic habitats can be circumvented by anthropogenic changes at the watershed scale. The routing of water from human development to streams via channelized flow, for example, can result in changes to the stream hydrograph, changes in channel morphology, reduced efficacy of the riparian pollutant removal function, and increases in fine sediment inputs to streams.

- More research is needed on practical ways to maintain or emulate natural disturbances in watersheds dominated by human land uses (e.g., agriculture, residential). This includes disturbances such as flooding and landslides that are essential for maintaining aquatic habitats.

9.3. Key Management Implications from Chapters 1 through 8

- Protection and restoration of riparian ecosystems continues to be critically important because: a) they are disproportionately important, relative to area, for aquatic species, e.g., salmon, and terrestrial wildlife, b) they provide ecosystem services such as water purification and fisheries (Naiman and Bilby 2001; NRC 2002; Richardson et al. 2012), and c) by interacting with watershed-scale processes, they contribute to the creation and maintenance of aquatic habitats.

- Stream riparian ecosystems encompass the riparian zone, the active floodplain including riverine wetlands, and the terraces and adjacent uplands that contribute matter and energy to the active channel or active floodplain (Gregory et al. 1991; Naiman and Bilby 1998).

Engineered large wood structure in the north fork of the Lewis River/ Chuck Stambaugh, WDFW

Such terraces and adjacent uplands are called the zone of influence.

- The width of the riparian ecosystem is estimated by one 200-year site-potential tree height (SPTH) measured from the edge of the active channel or active floodplain. Protecting functions within at least one 200-year SPTH is a scientifically supported approach if the goal is to protect and maintain full function of the riparian ecosystem.

- Where the riparian zone is narrow (<100 ft [30 m]) and the zone of influence lacks tall trees (<100 ft), (e.g., in parts of the Columbia Plateau Ecoregion) the pollution removal function may determine the width of the zone of influence.

- The riparian ecosystem begins at the edge of the active channel or active floodplain, whichever is wider. As the active channel moves back and forth across the channel migration zone (CMZ), the riparian ecosystem moves with it. Consequently, there are times when the riparian ecosystem lies adjacent to and immediately outside the CMZ. Hence, to maintain riparian ecosystem functions, management must anticipate and protect future locations of the riparian ecosystems.

- A near consensus of scientific opinion holds that the most effective and reliable means of maintaining viable self-sustaining fish, especially salmon, is to maintain/restore ecosystems to conditions that resemble or emulate their historical range of natural variability (Swanson et al. 1994; Reeves et al. 1995; Bisson et al. 2009). This opinion is based in part on the complexity of processes that affect the expression of habitats over time and space.

- The protection and restoration of watershed-scale processes, especially related to hydrology, water quality, connectivity, and inputs of wood, shade, and sediment are important for aquatic system function, and help maximize the ecological benefits of riparian ecosystem protections.

- Riparian areas and surrounding watersheds are complex and dynamic systems comprised of many interacting components. Natural disturbances (floods, fire, and landslides) across the watershed and through time create the mosaic of conditions necessary for self-sustaining populations of fish, especially salmon, and other aquatic organisms.

- Impending changes to aquatic systems as a result of climate change increase risk to species already threatened by human activities. The effects of climate change on rivers and streams threaten to reduce fish distribution and viability throughout the Pacific Northwest (Beechie et al. 2013).

- The use of the precautionary principle and adaptive management are particularly appropriate when dealing with complex and dynamic systems, and when we have uncertainty related to exactly how management activities affect functioning of watersheds and riparian ecosystems.

9.4. Width of Riparian Ecosystems

The width of the riparian ecosystem is based on the zone of influence, and the zone of influence is based on riparian ecosystem functions (or processes) that affect aquatic habitats. FEMAT's (1993) curves are conceptual models describing how four key riparian ecosystem functions change with distance from the stream channel (Figure 9.1). The four functions are root strength (i.e., streambank stability), stream shading, litter fall (i.e., input of detrital nutrients), and coarse wood debris to

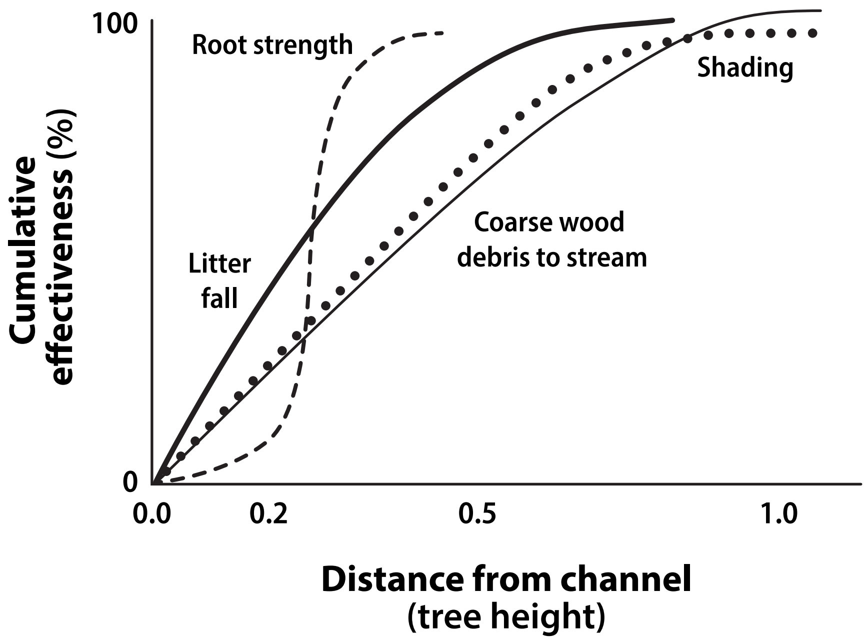


Figure 9.1. The "FEMAT Curves" (FEMAT 1993) are generalized conceptual models describing some riparian ecosystem functions and processes as the distance from a stream channel increases. "Tree height" in this figure is based on the average maximum height of the tallest dominant trees (200 years old or greater) and is referred to as site-potential tree height (SPTH).

stream (i.e., recruitment of large wood). FEMAT (1993) also included the pollution removal function (i.e., water quality) but did not create a curve for it. Originally, the curves were meant to convey two important points: 1) these four riparian ecosystem functions occur within one SPTH, and 2) the marginal return for each function decreases as distance from the stream channel increases. In other words, the FEMAT curves show that areas closer to the stream provide more of each function per unit width than areas farther from the stream.

The FEMAT curves are generalizations that ignore site-specific variability in riparian function among stream reaches. Nevertheless, FEMAT's conceptual models have influenced the width of riparian management zones (RMZs) in a number of important conservation plans such as the Forests and Fish Habitat Conservation Plan (DNR 2005).[2] Protecting natural vegetation within at least one SPTH is a scientifically supported approach if the goal is to protect and maintain ecological functions of riparian areas for the benefit of aquatic species. All else being equal, protecting areas narrower than one SPTH will result in lower levels of ecological function, and generally increase risk to fish and aquatic wildlife.

Although not all riparian functions are strongly associated with tree height (e.g., pollution removal), several key functions are, e.g., large wood recruitment, stream shading, and litter fall. Hence, the FEMAT curves use tree height of the dominant tree species to help estimate the amount of function provided by riparian areas of different widths. Consequently, the area within a site-potential tree height has often been used to define the extent of the "riparian ecosystem." However, FEMAT curves dealt strictly with functions strongly associated with vegetation and thus did not attempt to represent all aspects of the riparian ecosystem, such as other riparian functions or habitats for terrestrial species. For example, FEMAT curves did not explicitly address floodplains and riverine wetlands, sub-surface water movements, nutrient dynamics, pollutant removal, or groundwater recharge.

Given its utility, the height of site-potential trees has been described for a wide variety of tree species and can be readily found in the silvicultural literature. Fox (2003) found that mean heights of dominant trees in riparian old-growth forest of Washington range from 100 to 240 ft (30 to 70 m). The wide range of heights reflects differences in site productivity, i.e., local differences in soil nutrients and moisture, light and temperature regimes, and topography (Avery and Burkhart 1994: 278). Site productivity is described quantitatively through a site index, which is the average height that dominant trees of a particular species are expected to attain at a specified tree age. Tables (e.g., King 1966) have been developed to predict the future average height of dominant trees on a site.

FEMAT (1993:V-34) defined site-potential tree height as "the average maximum height of the tallest dominant trees (200 years or more) for a given site class." The key phrase in this definition is "200 years or more" which refers to the approximate minimum age of old-growth forests (Franklin and Spies 1991). This reflects FEMAT's underlying assumption that old-growth forest conditions are needed for full riparian ecosystem functions. Because Douglas-fir can continue height growth for more than 200 years (Herman and Lavender 1990), site-potential height based on age 200 years is the minimum width for full riparian ecosystem functions according to FEMAT. Moreover, in addition to changes in SPTH with increasing forest age, other riparian forest components (e.g., basal area of live trees, species composition, volume of dead woody material) continue to change after 200 years of age (Fox 2003) resulting in what may be important but relatively unstudied implications for riparian ecosystem functions and values.

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[2] However, we note that the Forests and Fish Habitat Conservation Plan (FFHCP) did not adopt the FEMAT (1993) definition of site-potential tree height. The FFHCP defined SPTH as the height of dominant trees at age 100 years.

Quinault River/Wendy Cole, WDFW

9.5. The Condition of Riparian Ecosystems

Many scientific experts (Swanson et al. 1994; Reeves et al. 1995; Bisson et al. 2009) believe that the most likely way to realize viable self-sustaining fish, especially salmon, populations is to maintain/restore ecosystems to conditions that resemble their historical range of natural variability[3] (hereafter historical conditions). The scientific rationale for this opinion is that fish species have over many millennia adapted to particular disturbance regimes and consequent habitat conditions. Therefore, many scientists believe that dramatic deviation from historical conditions reduces the average survival and fecundity of animal populations, and thus puts species at risk of extinction (Swanson et al. 1994).

We define the historical conditions as the dynamic range of ecosystem composition, structure, and functions prior to significant impacts by Euro-American settlement (Swanson et al. 1994). Landres et al. (1999) suggest that the historical conditions provides guidance, context, and benchmarks for management. Understanding changes to historical conditions imposed by past management action (e.g., clearcutting riparian areas) can inform decisions on future management and realistic future objectives, such as desired future condition (DFC). Articulating riparian DFC with awareness of historical conditions provides a meaningful benchmark for assessing potential loss of riparian function and potential risks to fish and wildlife populations (Landres et al. 1999; Morgan et al. 1994). Because historical composition and structure were driven by disturbances, Landres et al. (1999) also suggest that managers should maintain, or at least emulate, the disturbance processes that created historical conditions. In other words, historical conditions describe relationships between species and habitats and thus provide context and guidance for managing ecological systems in the future. Emulating historical conditions could include the appropriate use of engineered solutions (e.g., stormwater basins to reduce peak flows, placing large wood in streams) to protect or restore habitats. Bisson et al. (2009) promote this same concept for conserving freshwater habitats of anadromous salmonids.

Managing a site, reach, or watershed to more closely resemble historical conditions may also effectively serve as a coarse-filter conservation strategy. Coarse filter strategies are based on the idea that managing for a habitat type, perhaps by emulating historical

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[3] Also referred to as "historical natural variability," "natural variability," "range of natural variability," "natural range of variability," "historical range of variability," or "reference variability" (Landres et al. 1999).

Riparian ecosystem near Naches/Mary Huff, WDFW

conditions, will more efficiently conserve many species rather than managing for each species separately (Hunter et al. 1988; Swanson et al. 1994). Consistent with Landres et al. (1999), managing riparian areas with a goal of emulating historical conditions may be more cost effective in the long run than other approaches as it should require fewer investments in mitigation for degraded riparian ecosystem functions. In other words, managing for historical conditions is consistent with the idea that avoiding impacts to habitat is less expensive and more scientifically defensible than restoring habitat for each species after impacts.

Emulating an ecosystem's historical conditions poses both empirical and conceptual challenges. Empirical data needed to estimate the historical conditions are seldom available, and when available, the data are usually inadequate for the task of reliably estimating historical variation over time and space (Keane et al. 2009). The biggest conceptual challenge is posed by the ever-changing environment, much of it wrought by humans. While the historical conditions provides the best description of habitats that supported fish and wildlife species in the past, we recognize the challenge of applying that knowledge to watersheds that have been drastically altered by exotic species, fire suppression, intensive land uses, and climate change. Despite these challenges, Keane et al. (2009) still believe that the historical conditions provides an essential perspective for managers.

Historical conditions are the result of natural processes such as frequent floods, landslides, and wildfire – some of catastrophic magnitude – occurring throughout a watershed. Hence, for obvious reasons, managing riparian areas so that they roughly match their historical conditions may be unrealistic, especially in residential or agricultural settings. Furthermore, WDFW acknowledges that other uses of riparian areas as legitimate and worthwhile. Such uses and a wide variety of societal values must also be considered when developing strategies, plans, or rules for riparian areas. Therefore, rather than a goal, the historical conditions may better serve as a guide.

Dynamic mosaics of riparian and aquatic habitats are created, maintained, and altered in predictable (e.g., seasonally) and unpredictable ways (e.g., due to random events like storms, fires, and landslides).

9.6. Other Important Issues

We highlight three important issues that deserve more in-depth discussion: complexity, connectivity, and climate change. We also consider management implications of each issue for riparian areas and watersheds.

9.6.1 Complexity

The ecologist Frank Egler stated, "Ecosystems are not only more complex than we think, but more complex than we can think" (Egler 1977). Riparian ecosystems fit that description – they are dynamic, nonstationary, nonlinear systems comprised of many interacting components. In dynamic systems, conditions continually change over time. Dynamic equilibrium, for instance, means that conditions continually fluctuate about an average state, and the long-term statistics describing that process – mean, variance, fluctuation frequency – remain virtually constant. Nonstationary means that a system's functional relationships are changing over time, and consequently, the long-term statistics describing the process are changing. Some scientists believe that future climate change will lead to more pronounced nonstationarity in aquatic ecosystems (Milly et al. 2008). Nonlinear implies that functional relationships between ecosystem components are complicated. Under certain conditions, some types of nonlinear relationships may exhibit chaotic behavior. Not surprisingly, predicting the effects of management actions on dynamic, nonstationary, nonlinear systems is challenging.

Frissell et al. (1986) helped conceptualize watersheds as hierarchical systems where larger scale components, structures, and processes act primarily in the downstream direction on smaller scale components, structures, and processes. Factors that affect processes

The Nooksack River is located in the northwest part of Washington. It drains an area of the Cascade Range around Mount Baker, and flows through a fertile agricultural area before emptying into Bellingham Bay./Department of Natural Resources over the longest duration can be considered "drivers" of the conditions and processes at lower levels in the hierarchy. Climate, geology, topography (and land cover) largely determine watershed characteristics and processes (paraphrasing the framework of Montgomery 1999). However, the composition, structure, and functions of streams and riparian ecosystems do not simply arise from these drivers. They are created and modified by continuous and episodic fluxes of materials (particularly water, sediment, and wood) and energy, which are broadly termed watershed processes. Dynamic mosaics of riparian and aquatic habitats are created, maintained, and altered in predictable (e.g., seasonally) and unpredictable ways (e.g., due to random events like storms, fires, and landslides).

These types of nonlinear dynamics are intrinsic properties of all ecosystems and important for the creation and maintenance of habitats that fish and wildlife populations depend on (Fausch et al. 2002; Ward et al. 2002; Wiens 2002), including salmon (Merz et al. 2015). The availability of multiple types of habitat (i.e., habitat heterogeneity) within and among streams may be essential for the persistence of multiple life history strategies in salmon species (Hilborn et al. 2003; Bisson et al. 2009; Waples et al. 2009; Merz et al. 2015) and the coexistence of other aquatic and riparian-dependent species (Bellmore et al. 2015).

The availability of multiple types of habitat within and among streams may be essential for the persistence of multiple life history strategies in salmon species and the coexistence of other aquatic and riparian-dependent species.

How should we approach the management of ecosystem complexity? Because ecosystems are wickedly complex (Ludwig 2001) and exhibit irreducible natural variability at multiple spatial and temporal scales, managers should adopt an attitude of humility. Over-confident managers who approach decisions with resolute certainty may fail to anticipate problems or recognize potential risks. Humility should instill in managers a desire to understand more before acting and a further desire to continually improve one's understanding of the system. The former is a quality of the precautionary principle, and the latter is a philosophical underpinning of adaptive management. Adaptive management is discussed later in this chapter.

9.6.2. Connectivity

Several chapters in this volume introduce the idea of watershed connectivity, in particular, longitudinal, lateral, and vertical connectivity. Connectivity is vital to aquatic ecosystem function and especially important at conferring resiliency to the negative effects of climate change (see section 9.6.3). Streams and their associated riparian ecosystems transport water, wood, sediment, nutrients, and organisms predominantly, but not exclusively in the downstream direction. Longitudinal (upstream to downstream) fragmentation of streams, for example by road crossings that block passage of fish, can reduce habitat capacity for individual species and change aquatic community composition by impeding movements of fish and other animals. Dams and undersized culverts also reduce the transport of large wood and sediment from upstream areas, which can adversely impact habitat composition and structure. Loss of lateral connectivity between streams and their floodplains due to levees, dams, or channel incision can reduce development of off-channel aquatic habitats, alter riparian plant communities, and impact processes such as water storage and nutrient cycling. Reduction of vertical connectivity, i.e., exchanges between stream water and deeper groundwater in the hyporheic zone, can increase stream water temperatures, alter stream chemistry, and impact macroinvertebrate populations.

The importance of connectivity is also supported by our understanding of the extents at which aquatic species, especially fish, use stream habitat (Fausch et al. 2002). Movement of individuals among reaches and watersheds to locate specific habitat conditions (e.g., suitable water temperatures) is increasingly acknowledged as important to fish population persistence, particularly in light of climate change.

Bull trout in Deep Creek, Yakima County/Eric Anderson, WDFW

Maintaining connectivity in aquatic ecosystems likely requires management at site and watershed-scales. Management to protect or restore salmon and other aquatic species should ensure that all parts of a stream network that could be accessible to fish are in fact accessible to fish. That is, artificial structures should not impede fish movement. This requires watershed-scale stream network inventories and site-scale correction of artificial barriers. Additionally, barriers to the longitudinal or lateral transport of sediment and wood should be corrected, and hyporheic exchange should be enhanced by restoring the processes that supply in-stream large wood and coarse sediments. Where practical, management should ensure that the flows of water, sediment, and wood continue to flow as they did historically.

9.6.3. Climate Change

Climate change is expected to directly affect aquatic ecosystems by increasing air and water temperatures, altering stream hydrology, and impacting sediment yields throughout Washington. Indirect effects are too numerous to mention here but are likely to include changes to riparian plant communities, changes in fire regime, introduction of new diseases, exotic pests, and competitors with salmon. Because water temperature is widely recognized as one of the most important environmental factors influencing the geographic distribution, growth, and survival of fish (Regier et al. 1990; Armour 1991; McCullough 1999), the warming effects of climate change on rivers and streams threaten to reduce fish distribution and population viability (Beechie et al. 2013). The Pacific Northwest experienced a significant warming trend in summer stream temperatures of approximately 0.40°F/decade (0.22°C/decade) between 1980 and 2009 (Isaak et al. 2012). In the future, August stream temperatures are expected to increase an average of 5.1°F (2.8°C) by the 2080s (Isaak et al. 2015), and stream temperatures are expected to increase in all seasons of the year. The effects of climate change on surface water temperatures are compounded by other anthropogenic increases of water temperature caused by removal of riparian vegetation, water withdrawals, and channelization of streams (Poole and Berman 2001; Moore et al. 2005).

Climate change is projected to cause major changes in hydrology across Washington. Scientists have already detected negative trends in glacier volume and snowpack (Granshaw and Fountain 2006, Sitts et al. 2010, Stoelinga et al. 2010) and earlier peak streamflow in many rivers (Stewart et al. 2005). These trends are expected to continue in the future, along with increasing flood magnitudes, declining summer minimum flows, and rising stream temperatures (Elsner et al. 2010, Mantua et al. 2010). In Washington State, projected changes in future annual total precipitation are generally small compared to year-to-year fluctuations in seasonal and annual rainfall. Nonetheless, hydrological projections for the mid to late 21st century show a shift in flood frequencies that results in larger peak flows at all recurrence intervals , e.g., 2-year, 5-year, 10 year, etc. in some watersheds (Salathè et al. 2014). Changes are projected to be most pronounced in middle elevation basins, where a substantial proportion of the basin is located near the snowline (i.e., the “mixed rain and snow” zone). In these watersheds, warming will cause more precipitation to fall as rain instead of snow, which will decrease snow accumulation, hasten melt, and increase runoff (Hamlet et al. 2007).

Changes in stream flow are expected to alter sediment transport and channel morphology, however, published research analyzing the potential impact of future climate change on fluvial processes is limited. Praskievicz (2015) modeled the effects of future climate on geomorphic responses in three snow-dominated river basins of Idaho and eastern Washington. The results from the first site on the Tucannon River indicate that net sediment deposition is likely to occur, with increasing mid-channel bars. The second study site on the Coeur d’Alene River projects net erosion, and results for the third site forecast minimal changes on the Red River. These varying results indicate that the impacts of climate on sediment movement also depend on local context, i.e., how reach traits, such as substrate size or riparian vegetation, affect a stream channel’s morphological stability or lateral mobility.

Management to minimize direct detrimental effects of climate change on aquatic habitat includes: 1) increasing shade to streams by protecting or restoring riparian forests, 2) decreasing thermal sensitivity of surface waters by increasing summer low flows, 3) maintaining longitudinal, lateral, and vertical connectivity within watersheds, 4) monitoring and mitigating stream temperatures, especially when management actions are known to cause stream temperature increases, and 5) developing contingency plans for responding to climate related changes to water temperature, hydrology, or channel morphology. Because riparian areas can provide longitudinal habitat connectivity, link aquatic and terrestrial ecosystems, and create thermal refugia for wildlife, climate change makes the restoration of riparian areas especially important (Seavy et al. 2009).

9.7. Uncertainty and Management

Chapters 2 through 8 demonstrate that management of riparian ecosystems often involves substantial and unavoidable uncertainty, despite the fact that many riparian ecosystem functions are well understood. We include a discussion on uncertainty in management because effective use of scientific knowledge requires an understanding of its uncertainty (Fischhoff and Davis 2014) and because addressing uncertainty can lead to better long-term outcomes (Steel et al. 2009) for all stakeholders.

Some uncertainty is due to limited knowledge. For instance, the historical composition and structure of native riparian vegetation in the Columbia Plateau remains relatively unknown (see Chapter 7). Research can improve knowledge and reduce this type of uncertainty. A less well-appreciated type of uncertainty is associated with stochastic environmental variability, which introduces irreducible uncertainty or "unavoidable unpredictability" (sensu Spielgelhalter and

Riesch 2011) into the expected effects of management decisions. In other words, the longer one's time horizon for management decisions, the more likely that less expected, or even unexpected effects are going to occur. This type of uncertainty can be quantified through research, but never reduced. For example, large floods may unexpectedly alter channel form (e.g., widen a channel) and as a result change a stream's thermal regime. This in turn can decrease the effectiveness of management aimed at maintaining stream temperature based on pre-flood riparian conditions (see Chapter 4). This concept is important because conditions created by some land management actions are ubiquitous and long-lived (e.g., young forests after timber harvest), thus creating a situation where less expected effects due to environmental variability are virtually guaranteed.

Using stream temperatures to discuss uncertainty is instructive because our understanding of riparian effects on stream temperatures is arguably the best developed of any riparian function (see Chapter 4). We know for instance how certain ecosystem composition, structures, and functions (predictor variables) affect stream temperatures, so we could accurately predict the effects of common management actions with high certainty. Unfortunately, logistical or economic constraints of collecting predictor variable data usually preclude the development and use of such models. Instead, we often employ simpler, statistical models or correlations to inform management. For instance, we can generally predict changes to water temperatures based on changes to shade from riparian vegetation because changes in shade are often strongly correlated with changes in water temperatures (Cristea and Janisch 2007).

Figure 9.2 shows the results of several studies relating changes in summer stream temperatures to changes in riparian buffer width. Stream temperatures were measured before and after reducing the widths of forested riparian buffers, which decreased shading of streams. The results of such studies are often assumed to predict the effects of shade reduction on stream temperatures in other locations. However, because each stream is unique (e.g., differences in stream widths, topographic shade, groundwater flows, etc.), changes in water temperature based solely on shade are relatively uncertain. For example, Jackson et al. (2001) found that 30-ft wide forested buffers caused summer stream temperatures to increase by an average of 4.2°F (2.4°C), but the 95% confidence interval for the average was -0.1° to 8.6°F (0.0 to 4.8°C). At one site, temperatures decreased, likely due to unmeasured effects on groundwater flow (Janisch et al. 2012). Such uncertainty is common. Additional studies of this type might narrow confidence intervals and more accurately estimate the average response, but predictions of stream temperature based on buffer width alone will remain highly uncertain despite our knowledge that shade can strongly affect water temperature.

Similarly, empirical data describe wood recruitment from riparian areas as increasing with distance from the channel (i.e., riparian area width), but this relation varies widely among streams (Figure 9.3). Other potential predictor variables might include the wood recruitment mechanism, topography, fire regime, and forest age. However, we are unaware of other recruitment models that include variables that would allow us to predict the effects of management actions on wood recruitment with high certainty.

Currently, most riparian ecosystem functions described in this volume are best described with statistical models or correlations that often reliably predict the direction or sign (+ or -) of the correlation and some of the variability around the central tendency. Many such models rely on data from few sites that poorly describe the average or range of relations. For example, sample sizes for studies in Figure 9.2 are small, i.e., *N* = 6, 3, 5, and 5. Additional studies with larger sample sizes could better describe responses to shade reduction but are unlikely to substantially reduce uncertainty if they do not consider other important predictors in addition to buffer width.

Even as we conduct studies that reduce uncertainty associated with a lack of knowledge, we are continually challenged to make sound decisions into the future by irreducible uncertainty due to stochastic events, such as natural disturbances, and non-stationary processes like climate change. As riparian ecosystems change through time, whether due to future wildfire, floods, or climate change, we must ensure that our management strategies meet their intended goals. Uncertainty about the future effects of current management actions is perhaps the biggest challenge faced by managers because those effects of management often persist over long periods or may be practically irreversible (e.g., loss of riparian areas due to residential development). With this uncertainty comes the risk that the unpredictable may convert current and successful regulatory policies or management practices into future failures.

We can never eliminate uncertainty, but we can make management decisions that are appropriate given our current level of uncertainty. We recommend reducing risk to riparian ecosystems by: 1) reducing uncertainty through new research, 2) explicitly acknowledging and directly addressing uncertainty in policy and management decisions that affect fish and wildlife habitats, 3) considering the application of the precautionary principle to reduce unacceptable risk (O'Riordan and Jordan 1995; Gullet 1997). That is, erring on the side of caution, particularly when dealing with environmental impacts that are difficult to undo (irreversible damage or long recovery times), and 4) practicing adaptive management.

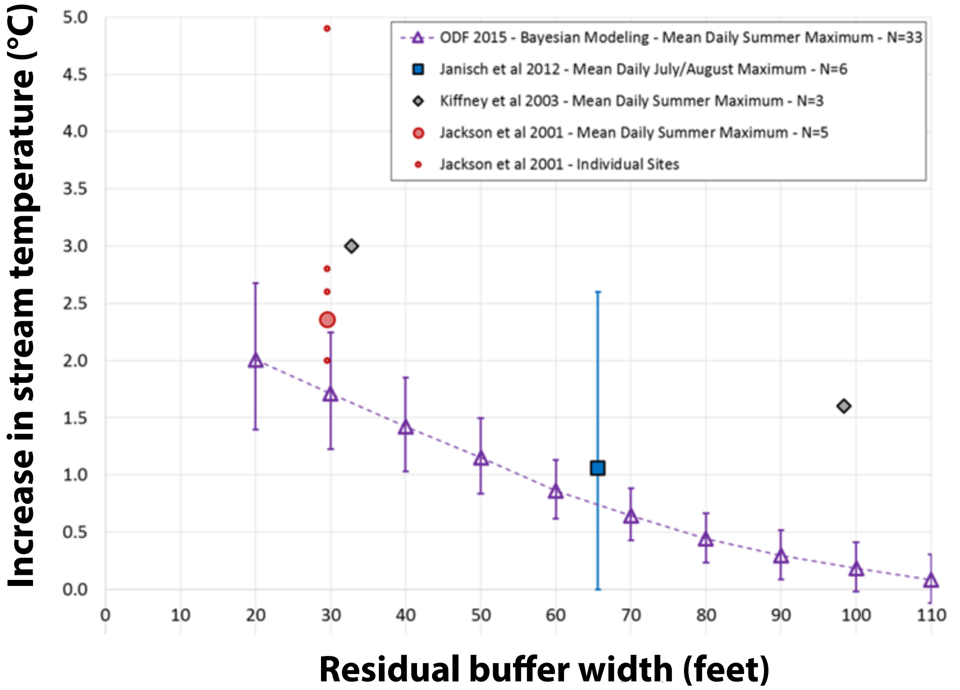

Figure 9.2. Observed temperature response associated with "no-cut" riparian buffers with adjacent clearcut harvest. Only studies that employed a Before-After-Control-Impact design and conducted in Pacific Northwest forests are included. Bayesian modeling results (and 90% credible intervals) were derived from data collected as part of Groom et al. (2011). A change of 1°C is equivalent to a 1.8°F change. Analyses provided by P. Leinenbach, USEPA Region 10 (See Chapter 4 for a more complete description of this figure).

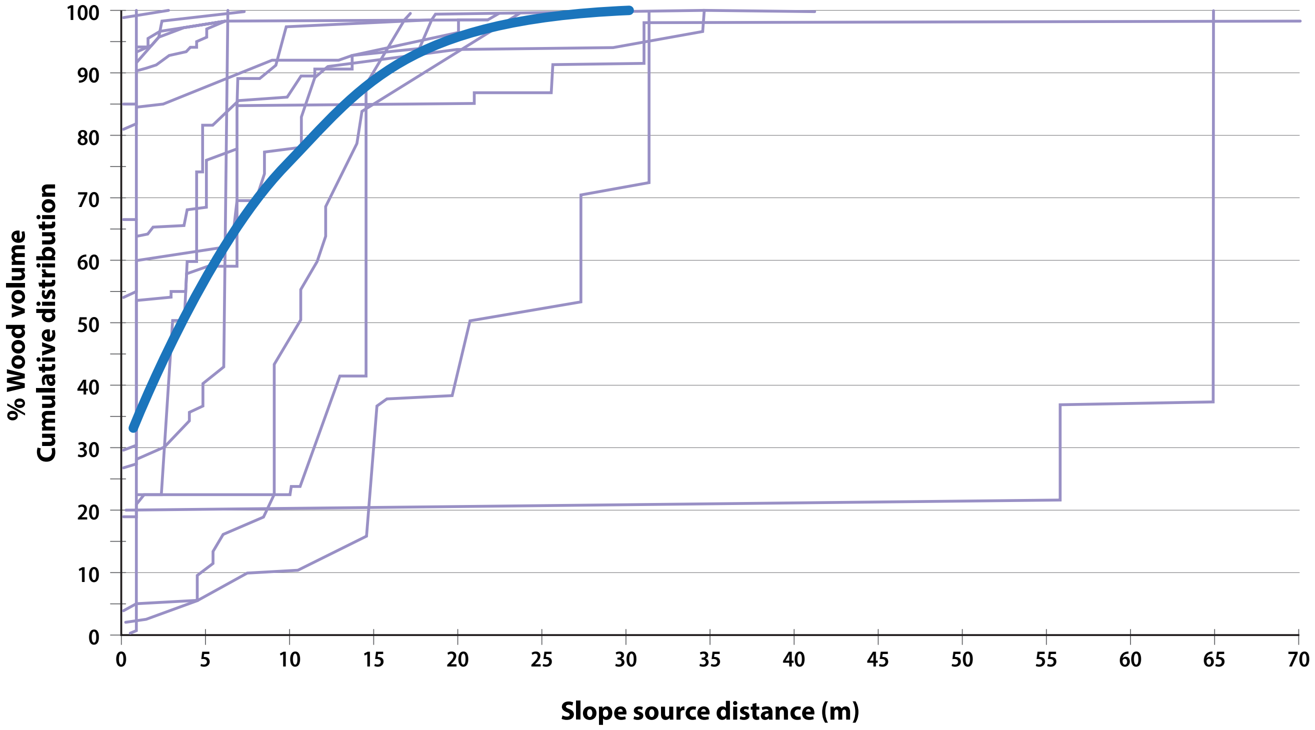

Figure 9.3. Sixteen large wood recruitment curves for second-growth redwood forest on 16 sites in northern California (from Benda et al. 2002). The theoretical prediction curve (thick line) is based on mortality recruitment only using random 360° fall trajectories. Curves to the left of the theoretical curve are sites where bank erosion is a major recruitment mechanism. Curves to the right of the theoretical curve are sites where landsliding is a major recruitment mechanism. (See Chapter 3 for a more complete description of this figure).

9.8. Adaptive Management

Despite new insights into ecosystems dynamics and better understanding of the effects of management actions on species and their habitats, natural resource management decisions will always struggle with some level of uncertainty. Adaptive management is widely recognized as a sensible, if not essential, approach to the management of natural resources under uncertainty. As originally conceived, adaptive management can be defined as the systematic acquisition and application of reliable information to improve management over time (Wilhere 2002). Reliable information is acquired through statistically-valid research and monitoring. Furthermore, adaptive management can facilitate a way forward when levels of uncertainty become high enough to threaten stakeholder agreement about environmental policies (Wilhere and Quinn 2018).

The adaptive management process is a continual cycle consisting of planning, action, monitoring, evaluation, and adjustment (Bormann et al. 1994, Wilhere 2002) (Figure 9.4). Monitoring and evaluation form a feedback loop that provides information for management decisions. However, feedback alone cannot fulfill the goal of adaptive management, that is, the continual improvement of management. Ideally, an "adaptive management plan" integrates every phase of management with monitoring, and considers how information collected through monitoring will lead to future changes in management. Developing an adaptive management plan should be a collective endeavor of policy makers, managers, field staff, and scientists.

Successful adaptive management requires an understanding of all stakeholder perspectives. Hence, developing an adaptive management plan should begin with discussions amongst stakeholders about their concerns regarding uncertainty and risk. The plan should be developed through a negotiated process, and it should explain how specific risks will be addressed through a program of systematic learning (e.g., management experiments, focused monitoring, targeted literature reviews, etc.). Importantly, the best adaptive management plans have agreements about the specific information necessary to change policy or management and what those changes might be.

Adaptive management programs can fall victim to the widespread confusion between the roles of values and science in management decisions (Wilhere and Quinn 2018). Maintaining a clear distinction between values (cultural, social, and economic) and science, and between the roles of policy makers and scientists, is critical to success. That said, WDFW recognizes that natural resource management or land use policies must often balance competing ecological, economic, and social values. In fact, one of WDFW's conservation principles is that management decisions should integrate ecological, economic, and social perspectives. This volume addresses only the ecology of riparian area management; it does not cover economic and social aspects. Determining an acceptable balance requires a political process involving stakeholders, local governments, tribes, and state agencies. We hope this volume can help to inform that process.

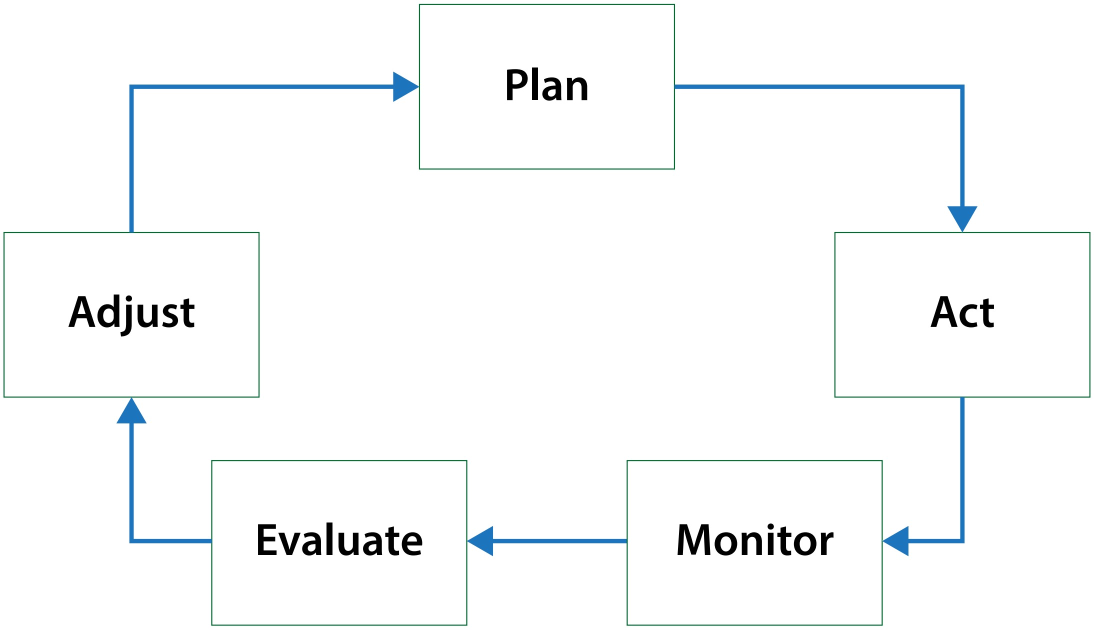

Figure 9.4. The adaptive management cycle (modified from Bormann et al. 1994).

9.9. Literature Cited[4]

Armour, C. 1991. Guidance for evaluating and recommending temperature regimes to protect fish. Biological Report 90(22). Fish and Wildlife Service, Washington, D.C. (viii)

Avery, T.E., and H.E. Burkhart. 1994. Forest measurements. McGraw-Hill, New York. (i)

Beechie, T.J., H. Imaki, J. Greene, A. Wade, H. Wu, G. Pess, P. Roni, J. Kimball, J. Stanford, P. Kiffney, and N. Mantua. 2013. Restoring salmon habitat for a changing climate. River Research and Applications 29:939–960. (i)

Bellmore, J.R., C.V. Baxter, and P.J. Connolly. 2015. Spatial complexity reduces interaction strengths in the meta-food web of a river floodplain. Ecology 96:274-283. (i)

Benda, L.E., P. Bigelow, and T.M. Worsley. 2002. Recruitment of wood to streams in old-growth and second-growth redwood forests, northern California, U.S.A. Canadian Journal of Forest Research 32:1460-1477. (i)

Bisson, P.A., J.B. Dunham, and G.H. Reeves. 2009. Freshwater ecosystems and resilience of Pacific salmon: habitat management based on natural variability. Ecology and Society 14(1):45. (i)

Bormann, B.T., P.G. Cunningham, M.H. Brookes, V.W. Manning, and M.W. Collopy. 1994. Adaptive ecosystem management in the Pacific Northwest. PNW-GTR-341. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Cristea, N., and J. Janisch. 2007. Modeling the effects of riparian buffer width on effective shade and stream temperature. Washington Department of Ecology, Olympia. (viii)

Cummins, K.W. 1974. Structure and function of streams ecosystems. BioScience 24:631-641. (i)

DNR (Washington Department of Natural Resources). 1999. Forests and fish report. Forest Practices Division, Olympia, Washington. (viii)

DNR (Washington Department of Natural Resources). 2005. Final forest practices habitat conservation plan. Forest Practices Division, Olympia, Washington. (viii)

Egler, F.E. 1977. The nature of vegetation: its management and mismanagement. Aton Forest, Norfolk, Connecticut. (viii)

Elsner, M.M., L. Cuo, N. Voisin, J.S. Deems , A.F. Hamlet, J.A. Vano , K.E.B. Mickelson, S.Y. Lee, and D.P. Lettenmaier. 2010. Implications of 21st century climate change for the hydrology of Washington State. Climatic Change 102:225–260. (i)

Fausch, K.D., C.E. Torgersen, C.V. Baxter, and H.W. Li. 2002. Landscapes to riverscapes: bridging the gap between research and conservation of stream fishes. BioScience 52:483-498. (i)

FEMAT (Forest Ecosystem Management Assessment Team). 1993. Forest ecosystem management: an ecological, economic and social assessment. U.S. Department of Agriculture and U.S. Department of the Interior, Portland, Oregon. (viii)

Fischhoff, B., and A.L. Davis. 2014. Communicating scientific uncertainty. Proceedings of the National Academy of Science 111(Supplement 4): 13664-13671. (i)

Fox, M.J. 2003. Spatial organization, position, and source characteristics of large woody debris in natural systems. Doctoral dissertation. University of Washington, Seattle. (i)

Franklin, J.F., and T.A. Spies. 1991. Composition, function, and structure of old-growth Douglas-fir forests. Pages 71-82 *in* L.F. Ruggerio, K.B. Aubry, A.B. Carey, and M.H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. PNW-GTR-285. U.S. Forest Service, Pacific, Northwest Research Station, Portland, Oregon. (viii)

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream classification: viewing streams in a watershed context. Environmental Management 10:199-214. (i)

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[4] References are classified according to RCW 34.05.271 (see Appendix 1) denoted by a lower case roman numeral (i – viii) in parentheses at the end of the citation.

Granshaw, F.D., and A.G. Fountain. 2006. Glacier change (1958–1998) in the North Cascades National Park Complex, Washington, USA. Journal of Glaciology 52:251-256. (i)

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones: focus on links between land and water. Bioscience 41:540-551. (i)

Groom, J.D., L. Dent, and J. Madsen. 2011. Stream temperature change detection for state and private forests in the Oregon Coast Range. Water Resources Research 47:1-12. (i)

Gullet, W. 1997. Environmental protection and the "precautionary principle": a response to scientific uncertainty in environmental management. Environmental Planning and Law Journal 14:52-69. (i)

Hamlet, A. F., P.W. Mote, M.P. Clark, and D.P. Lettenmaier. 2007. Twentieth-century trends in runoff, evapotranspiration, and soil moisture in the western United States. Journal of Climate 20:1468-1486. (i)

Herman, R.K., and D.P. Lavender. 1990. *Pseudotsuga menziesii*. Pages 527-540 *in* R.M. Burns and B.H. Honkala, technical coordinators. Silvics of North America volume 1: conifers. Agriculture Handbook 654. U.S. Forest Service, Washington, D.C. (viii)

Hilborn, R. 1987. Living with uncertainty in resource management. North American Journal of Fisheries Management 7:1-5. (i)

Hilborn, R., T.P. Quinn, D.E. Schindler, and D.E. Rogers. 2003. Biocomplexity and fisheries sustainability. Proceedings of the National Academy of Sciences 100:6564-6568. (i)

Hunter, M.L. Jr., G.L Jacobson Jr., and T. Web III. 1988. Paleoecology and the coarse filter approach to maintaining biological diversity. Conservation Biology 2:375-385. (i)

Independent Scientific Group. 1999. Scientific issues in the restoration of salmonid fishes in the Columbia River. Fisheries 24:210-219. (i)

Isaak, D., S. Wenger, E. Peterson, J. Ver Hoef, C. Luce, S. Hostetler, J. Dunham, J. Kershner, B. Roper, D. Nagel, D. Horan, G. Chandler, S. Parkes, and S. Wollrab. 2015. Development and application of NorWeST stream temperature climate scenarios for the Pacific Northwest. Webinar, January 27. North Pacific Landscape Conservation. Cooperative, Lacey, Washington. (viii)

Isaak, D.J., S. Wollrab, D. Horan, and G. Chandler. 2012. Climate change effects on stream and river temperatures across the northwest U.S. from 1980 - 2009 and implications for salmonid fishes. Climate Change 113:499-524. (i)

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber harvest impacts on small headwater stream channels in the coast ranges of Washington. Journal of the American Water Resources Association 37:1533–1549. (i)

Janisch, J.E., S.M. Wondzell, and W.J. Ehinger. 2012. Headwater stream temperature: interpreting response after logging, with and without riparian buffers, Washington, USA. Forest Ecology and Management 270:302-313. (i)

Keane, R.E., P.F. Hessburg, P.B. Landres, and F.J. Swanson. 2009. The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management 258:1025-1037. (i)

Kiffney, P.M., J.S. Richardson, and J.P. Bull. 2003. Responses of periphyton and insect consumers to experimental manipulation of riparian buffer width along headwater streams. Journal of the American Water Resources Association 40:1060-1076. (i)

King, J.E. 1966. Site index curves for Douglas-fir in the Pacific Northwest. Weyerhaeuser Forestry Paper No. 8. Forestry Research Center, Weyerhaeuser Company, Centralia, Washington. (viii)

Landres, P.B., P. Morgan, and F.J. Swanson. 1999. Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications 9:1179-1188. (i)

Ludwig, D. 2001. The era of management is over. Ecosystems 4:758–764. (i)

Mantua, N., I. Tohver, and A. Hamlet. 2010. Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. Climatic Change 102:187-223. (i)

McCullough, D.A. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids, with special reference to Chinook salmon. EPA 910-R-99e010. U.S. Environmental Protection Agency, Region 10, Seattle, Washington. (viii)

Merz, J.E., D.G. Delaney, J.D. Setka, and M.L. Workman. 2015. Seasonal rearing habitat in a large Mediterranean-climate river: management implications at the southern extent of Pacific salmon (*Oncorhynchus* spp.). River Research and Applications 32:1220-1231. (i)

Milly, P.C.D., J. Betancourt, M. Falkenmark, R.M. Hirsch, Z.W. Kundzewicz, D.P. Lettenmaier, and R.J. Stouffer. 2008. Stationarity is dead: whither water management? Science 319: 573-574. (i)

Montgomery, D.R. 1999. Process domains and the river continuum. Journal of the American Water Resources Association 35:397-410. (i)

Moore, R.D., D.L. Spittlehouse, and A. Story. 2005. Riparian microclimate and stream temperature response to forest harvesting: a review. Journal of the American Water Resource Association 41:813-834. (i)

Morgan, P., G.H. Aplet, J.B. Haufler, H.C. Humphries, M.M. Moore, and W.D. Wilson. 1994. Historical range of variability: a useful tool for evaluating ecosystem change. Journal of Sustainable Forestry 2:87-111. (i)

Naiman, R.J., and R.E. Bilby, editors. 1998. River ecology and management: lessons from the Pacific Coastal Ecoregion. Springer-Verlag, New York. (i)

Naiman, R.J., R.E. Bilby, and P.A. Bisson. 2000. Riparian ecology and management in the Pacific Coastal Rain Forest. BioScience 50:996-1011.

NRC (National Research Council). 2002. Riparian areas: functions and strategies for management. National Academy Press. Washington, D.C. (i)

O'Riordan, T., and A. Jordan. 1995. The precautionary principle in contemporary environmental politics. Environmental Values 4:191-212. (i)

Poole, G.C., and C.H. Berman. 2001. An ecological perspective on in-stream temperature: natural heat dynamics and mechanisms of human-caused thermal degradation. Environmental Management 27:787-802. (i)

Praskievicz, S. 2015. A coupled hierarchical modeling approach to simulating the geomorphic response of river systems to anthropogenic climate change. Earth Surface Processes and Landforms 40:1616-1630. (i)

Reeves, G.H., L.E. Benda, K.M. Burnett, P.A. Bisson, and J.R. Sedell. 1995. A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionary significant units of anadromous salmonids in the Pacific Northwest. American Fisheries Society Symposium 17:334-349. (i)

Regier, H.A., J.A. Holmes, and D. Pauly. 1990. Influence of temperature changes on aquatic ecosystems: an interpretation of empirical data. Transactions of the American Fisheries Society 119:374-389. (i)

Richardson, J.S., R.J. Naiman, F.J. Swanson, and D.E. Hibbs. 2005. Riparian communities associated with Pacific Northwest headwater streams: assemblages, processes, and uniqueness. Journal of the American Water Resources Association 41:935-947. (i)

Salathé Jr., E.P., A.F. Hamlet, C.F. Mass, S.-Y. Lee, M. Stumbaugh, and R. Steed. 2014. Estimates of twenty-first-century flood risk in the Pacific Northwest based on regional climate model simulations. Journal of Hydrometeorology 15:1881-1899. (i)

Seavy, N.E., T. Gardali, G.H. Golet, F.T. Griggs, C.A. Howell, R. Kelsey, S.L. Small, J.H. Viers, and J.F. Weigand. 2009. Why climate change makes riparian restoration more important than ever: recommendations for practice and research. Ecological Restoration 27:330-338. (i)

Sitts D., A.G. Fountain, and M. Hoffman. 2010. Twentieth century glacier change on Mount Adams, Washington, USA. Northwest Science 84:378-385. (i)

Spiegelhalter, D.J., and H. Reisch. 2011. Don't know, can't know: embracing deeper uncertainties when analysing risks. Philosophical Transactions of the Royal Society A 369:4730–4750. (i)

Stanford, J.A., J.V. Ward, W.J. Lis, C.A. Frissell, R.N. Williams, J.A. Lichatowich, and C.C. Coutant. 1996. A general protocol for restoration of regulated rivers. Regulated Rivers: Research & Management 12:391-414. (i)

Steel, E.A., T.J. Beechie, M.H. Ruckelshaus, A.H. Fullerton, P. McElhany, and P. Roni. 2009. Mind the gap: uncertainty and model communication between mangers and scientists. American Fisheries Society Symposium 71:357–372. (i)

Stewart, I.T., D.R. Cayan, and M.D. Dettinger. 2005. Changes toward earlier streamflow timing across western North America. Journal of Climate 18:1136-1155. (i)

Stirling, A., and D. Gee. 2003. Science, precaution, and practice. Public Health Reports 117:521-533. (i)

Stoelinga, M.T., M.D. Albright, and C.F. Mass. 2010. A new look at snowpack trends in the Cascade Mountains. Journal of Climate 23:2473-2491. (i)

Suter, G.W., L.W. Barnthouse, and R.V. O'Neill. 1987. Treatment of risk in environmental impact assessment. Environmental Management 11:295-303. (i)

Swanson, F.J., J.A. Jones, D.O. Wallin, and J.H. Cissel. 1994. Natural variability-implications for ecosystem management. Pages 80-94 *in* M.E. Jensen and P.S. Bourgeron, technical coordinators. Volume II: ecosystem management: principles and applications. PNW-GTR-318. U.S. Forest Service, Pacific Northwest Research Station, Portland, Oregon. (viii)

Thomas, J.W., technical editor. 1979. Wildlife habitats in managed forests the Blue Mountains of Oregon and Washington. Agriculture Handbook No. 553. U.S. Forest Service, Washington, D.C. (viii)

Walker, B., C.S. Holling, S.R. Carpenter, and A. Kinzig. 2004. Resilience, adaptability and transformability in social–ecological systems. Ecology and Society 9(2):5. Available: http://www.ecologyandsociety.org/vol9/iss2/art5. (January 2019) (i)

Waples, R.S., T. Beechie, and G.R. Pess. 2009. Evolutionary history, habitat disturbance regimes, and anthropogenic changes: what do these mean for resilience of Pacific salmon populations? Ecology and Society 14(1):3. Available: http://www.ecologyandsociety.org/vol14/iss1/art3/. (January 2019) (i)

Ward, J.V., K. Tockner, D.B. Barscott, and C. Claret. 2002. Riverine landscape diversity. Freshwater Biology 47:517–539. (i)

Wiens, J.A. 2002. Riverine landscapes: taking landscape ecology into the water. Freshwater Biology 47:501-515. (i)

Wilhere, G.F. 2002. Adaptive management in habitat conservation plans. Conservation Biology 16:20-29. (i)

Wilhere, G.F., and T. Quinn. 2018. How wide is wide enough?: values, science, and law in riparian habitat conservation. Natural Resources Journal 58:279-318. (i)

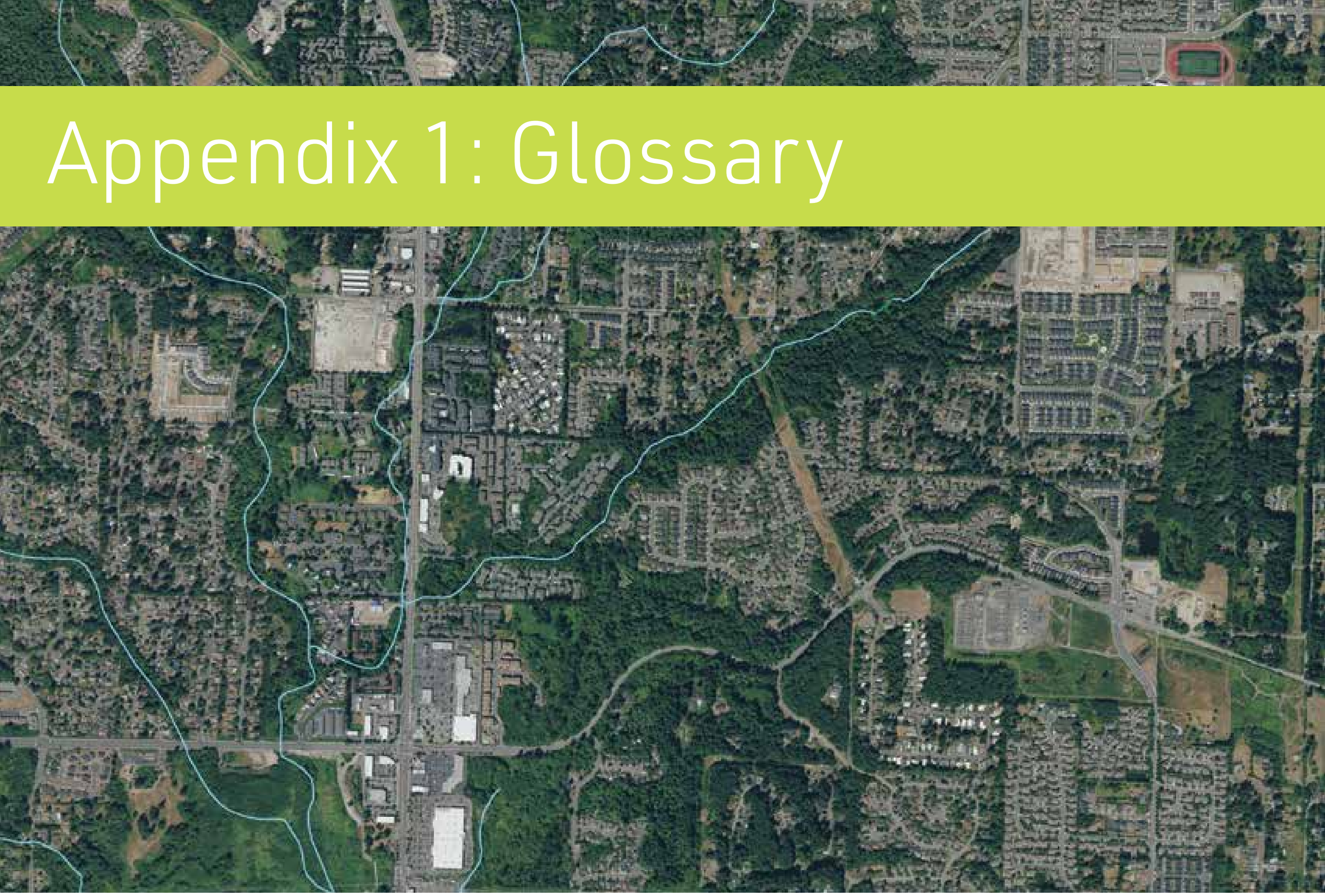
Wohl, E. 2005. Compromised rivers: understanding historical human impacts on rivers in the context of restoration. Ecology and Society 10(2):2. Available: http://www.ecologyandsociety.org/vol10/iss2/art2. (January 2019) (i)

Wynne, B. 1992. Uncertainty and environmental learning: reconceiving science and policy in the preventive paradigm. Global environmental change 2:111-127. (i)

Zar, J.H. 1996. Biostatistical analysis. Prentice-Hall, Upper Saddle, New Jersey. (viii)

Grande Ronde River/Alan Bauer

Appendix 1: Glossary



Riparian areas along streams (blue lines) in north King County/Terry Johnson, WDFW

Active channel

The portion of a stream channel defined by the lower limit of continuous riparian vegetation, and it may be delineated by absence of both moss on rocks and rooted vegetation. The upper most elevation of the active channel is sometimes equated with the ordinary high water mark.

Active floodplain

A portion of floodplain that is located between the active channel and adjacent terrace or hillslopes. Depending on the watershed, the flood return interval of the active floodplain varies between 1 and 10 years.

Adaptive management

The systematic acquisition and application of reliable information to improve management over time. Ideally, adaptive management treats management decisions as experiments in order to address critical uncertainties and learn more quickly from experience. It involves setting targets, monitoring outcomes, and adjusting management decisions based on results. Hallmarks of a well conceived adaptive management program include: 1) adequate funding for research, 2) a willingness to change course when pre-established triggers are reached, and 3) a commitment to monitor and evaluate conditions at appropriate spatial and temporal scales. See Ecosystem-based management.

Anthropogenic

Caused by human activity.

Channel confinement

A description of how much a channel can laterally move within its valley determined by the ratio of valley width (distance between toe of hillslopes on both sides of a valley) to active channel width. Typically, a segment is considered confined when the ratio is less than two and unconfined when greater than four.

Channel migration zone

The area within which a river channel is likely to move laterally over a specified period (e.g., 100 years).

Channel morphology

A stream channel's shape, which changes over time due to the interplay of water, sediments, vegetation, and certain animals, e.g., Beaver *Castor canadensis* and salmon.

Channel reach (stream)

A continuous segment of a channel that has similar physical features, such as gradient and confinement.

Channel slope or gradient

The average steepness of a stream segment measured as its change in elevation divided by its length. Typically, a segment's gradient is considered low if less than 2%, moderate between 2% and 4%, and high if greater than 4%.

Composition: See *Ecological composition*.

Disturbance regime

The frequency, magnitude, and duration of disturbance events.

Disturbance

A disturbance is any relatively discrete event in time that disrupts the composition, structure, or functions of ecosystems, communities, or populations.

Dynamic equilibrium

An ecological system's long-term state of relative stability that is maintained through opposing processes. Understanding an ecosystem's equilibrium, is a goal of ecosystem-based management.

Ecological (biological) integrity

Capacity of an ecological system to maintain a community of organisms that has species composition, diversity, and functional organization comparable to those of natural habitats within a region. An ecological system has integrity when its dominant ecological characteristics (e.g., composition, structure, function) occur within their historical ranges of natural variation.

Ecological composition

A term encompassing all parts of an ecosystem that include both living (biotic) and nonliving (abiotic) parts. Ecosystem composition is an important consideration in conservation.

Ecological function(ing)

1) The process or cause-effect relationships between two or more interacting ecosystem components; 2) the sum of processes that sustain the system; or 3) the capacity of natural processes to provide goods and services that satisfy human needs, either directly or indirectly.

Ecological process

Interactions among components of an ecosystem, both biotic (living organisms) and abiotic (chemical and physical) components. Many processes involve the transfer, conversion, or storage of matter and energy. This term is synonymous with the first definition of *ecological function*.

Ecological structure

The arrangement of and relations among the parts or components of an ecosystem.

Ecosystem

A spatially explicit unit of the Earth that includes all of the organisms, along with all components of the abiotic environment. Ecosystems have *composition, structure,* and *functions*.

Ecosystem-based management (EBM)

An integrated management approach that recognizes the full array of interactions within an ecosystem, including humans, rather than considering single issues, species, or ecosystem services in isolation. EBM often involves balancing ecological, economic, and social trade-offs within the context of existing laws and policies.

Erosion

The loosening and transport of soil particles and other sediment by water. Terrestrial erosion includes raindrop splash erosion, overland flow sheet erosion, surface flow rill (shallow) and gully (deeper) erosion. Channel erosion includes streambank erosion and channel incision. Rill and gully erosion diminishes the capacity of riparian areas to trap sediment and pollutants and often can be avoided with intact riparian vegetation.

FEMAT curve

A conceptual model that describes the relationship between various riparian *ecosystem functions* and distance from channel. The model consists of generalized curves that show the cumulative effectiveness of litter fall, root strength, shading, and coarse wood debris recruitment to stream as a function of distance from channel.

Flow regime

The distribution of stream discharges through space and time. Flow regimes can be described by their magnitude (e.g., mean annual, hourly maximum), timing, frequency or return periodicity, duration, spatial distribution, and rate of change. The pathways that water takes to reach a stream (e.g., surface runoff) exert a strong influence on the flow regime.

Function

Physical, chemical, or biological processes that occur within an ecosystem. See *ecological function(ing)* and *ecological process.*

Historical conditions: See *Range of natural variability.*

Hot moments

Periods of elevated denitrification rates. Hot moments can occur during a rainfall event. Also see *hot spots.*

Hot spots

Areas that exhibit high denitrification rates. Hot spots often occur in floodplains and other riparian areas with oscillating groundwater levels and/or higher hyporheic flows. Locations of hot spots can vary through time.

Hydrology

Study of the occurrence, distribution, movement, and properties of the water and their relationship with land.

Hyporheic zone

The area beneath the bed of a river or stream that is saturated with water.

Impervious surface

Ground surfaces that resist or prevent water infiltration, e.g., roofs of houses and roadways.

Incision

The process of downcutting into a stream channel leading to a decrease in the channel bed elevation. Incision is often caused by a decrease in sediment supply or an increase in stream flows capable of transporting (scouring) sediment.

Infiltration

The rate or process by which water on the ground surface enters the soil.

Macroinvertebrates

Animals, including insects, mollusks, crustaceans, and worms, that do not have a backbone, and are large enough to be seen without a microscope. Aquatic macroinvertebrates are commonly used as an indicator of habitat and water quality.

Mass wasting

The down slope movement of soil, sediments, and rock due to gravity (rather than water, wind, or ice, for example).

Nutrient cycling

The movement, uptake, transformation, storage, and release of nutrients, especially carbon, nitrogen, and phosphorus. Riparian characteristics that affect nutrient cycling include flow path, vegetation composition and structure, topography, groundwater level, and soil type.

Nutrient spiraling length

The distance nutrients move downstream during a complete cycle; a measure of nutrient utilization to nutrient supply. Long spiraling lengths indicate that the system is saturated with nutrients and organisms can no longer use the incoming nutrient loads. Streams in Washington forests typically have relatively short nutrient spiraling lengths.

Ordinary high water mark

This mark occurs along stream channels where the presence and action of waters are so common and usual, and so long continued in all ordinary years, as to mark upon the soil a character distinct from that of the abutting upland.

Precautionary principle

Erring on the side of causing no negative impact to natural resources when faced with uncertainty, especially for harm that is essentially irreversible. The precautionary approach involves: 1) taking preventive action (avoiding impacts), 2) shifting the burden of proof to the project proponents, 3) exploring a wide range of potential alternatives, and/or, 4) including multiple stakeholders and disciplines in decision making.

Process: See *Ecological process.*

Range of natural variability (or historical range of natural variability)

Refers to the range of ecological conditions (composition, structure, and functions) in a time period before widespread anthropogenic changes.

Riparian

An adjective meaning adjacent to a water body: stream, river, lake, pond, bay, sea, ocean. Riparian areas are sometimes referred to by different names: riparian ecosystems, riparian habitats, riparian corridors, or riparian zones. Depending on the context, these different terms may have somewhat different meanings.

Riparian buffer

Buffer refers to its purpose, which is to reduce or prevent adverse impacts to water quality, fisheries, and aquatic biodiversity from human activities occurring upslope of the buffer. Riparian buffers may also be called a riparian management zone. Riparian buffers managed specifically for pollutant removal may also be called a vegetated filter strip.

Riparian ecosystem

Riparian ecosystems are transitional between terrestrial and aquatic ecosystems and are distinguished by gradients in biophysical conditions, ecological processes, and biota. They are areas through which surface and subsurface hydrology connect waterbodies with their adjacent uplands. They include those portions of terrestrial ecosystems that significantly influence exchanges of energy and matter with aquatic ecosystems (i.e., a zone of influence). Our definition of riparian ecosystem does not include adjacent waters (i.e., river or streams, but does include riverine wetlands) and recognizes the riparian zone as a distinctive area within riparian ecosystems.

Riparian management zone

Riparian management zone is often synonymous with riparian buffer. See *Riparian buffer.*

Riparian zone

A distinctive area within riparian ecosystems. The riparian zone contains wet or moist soils and plants adapted to growing conditions associated with periodically saturated soils.

Riverscape

A landscape organized around a river system that includes the river network and contributing watershed along with other components that are not organized by watershed boundaries such as wildfire, mobile organisms, and wind-borne seeds. Distinct from uplands, it is primarily organized in a downstream direction (e.g., movement of water, sediment, and wood) but also contains lateral elements (e.g., floodplain interaction), vertical elements (e.g., interaction of surface and hyporheic flow), and upstream elements (e.g., migrating salmon).

Salmonid

A family of fish comprised of salmon, trout, and whitefish. Native salmonid species in Washington State include: Chinook Salmon *Oncorhynchus tshawytscha,* Chum Salmon *O. keta,* Coho Salmon *O. kisutch,* Pink Salmon *O. gorbuscha,* Sockeye Salmon/Kokanee *O. nerka*, Steelhead/Rainbow Trout *O. mykiss,* Cutthroat Trout *O. clarki,* Bull Trout *Salvelinus confluentus,* Dolly Varden *Salvelinus malma,* Pygmy Whitefish *Prosopium coulteri,* and Mountain Whitefish *P. williamsoni.* This list does not include names of subspecies.

Site class

The classification of a site's productivity based on the growth of the dominant tree species. Site classes vary based on local differences in soil nutrients and moisture, light and temperature regimes, and topography. Site classes are typically described as most productive (I) through least productive (V).

Site-potential tree height

The average maximum height of the tallest dominant trees for a given age and site class.

Stochastic process

An event involving chance or probability (e.g., landslide, flood). Stochastic processes may have patterns that can be analyzed statistically but cannot be precisely predicted.

Stream order

A hierarchical stream classification system in which headwater tributaries are classified as first order; when two first order tributaries meet they form a second order tributary, when two second order tributaries meet they form a third order tributary, and so on. Low order (1st-3rd) streams make up ~88% of the state's stream miles. Below its confluence with the Snake and Yakima rivers, the Columbia River is a 10th order river.

Structure: See *Ecological structure*.

Thermal loading potential

The potential amount of solar radiation (sunlight) available at a given location. Primary factors include shading (topographic and vegetative), latitude, elevation, and date.

Thermal regime (stream)

The distribution of stream temperatures through space and time. Thermal regimes can be described by their magnitude (e.g., monthly mean, hourly maximum), timing, frequency, duration, spatial distribution, and rate of change.

Thermal sensitivity (stream reach)

The susceptibility of a stream reach to changes in temperature. Thermal sensitivity typically increases with less stream flow, less groundwater input, and a wider channel to depth ratio.

Uncertainty (scientific)

The absence of information about something. Sources of uncertainty include, but are not limited to: natural variation (i.e., because ecosystems vary in unpredictable ways through time and space), model uncertainty (e.g., we do not understand a process), systematic error (e.g., poorly designed experiments), or measurement error.

Vegetative filter strips

A riparian buffer designed to capture nutrients, contaminant compounds, and sediment transported by run-off. Filter strips are sometimes synonymous with riparian buffers.

Water quality

Physical, chemical, and biological characteristics of water that describe its suitability to meet human needs or habitat requirements for fish and wildlife.

Watershed processes

The fluxes of energy (e.g., chemical, sunlight, wildfire) and materials (particularly water and sediment) that interact with biota (e.g., vegetation, salmon, Beavers, soil microbes) to form a watershed's physical features and characteristics, which give rise to its instream physical and ecological conditions. These processes occur within a context that reflects the watershed's climate, geology, topography, and existing human land use. Also see *Ecosystem process*.

Watershed

An area of land that drains to a common waterbody.

Wood recruitment

The process of wood moving from a riparian area to the stream channel. Sources of recruitment include bank erosion, windthrow, landslides, debris flows, snow avalanches, ice storms, Beaver, and tree mortality due to fire, insects, or disease.

Northwestern salamander/WDFW

Appendix 2: Categorization of Information Sources

Legal Requirement: RCW 34.05.271

The Revised Code of Washington (RCW) 34.05.271 requires the Washington Department of Fish and Wildlife (WDFW) to categorize sources of information used to inform technical documents that directly support implementation of a state rule or statute. Because WDFW Priority Habitat and Species documents—such as this one—are referenced in regulations for the Growth Management Act [e.g., WAC 365-190-130(4)] and Shoreline Management Act [e.g., WAC 173-26-221(5)(b)], we classify all references in the literature cited sections of Volume 1 into the following RCW 34.05.271 categories:

(i) Independent peer review: Review is overseen by an independent third party;
(ii) Internal peer review: Review by staff internal to the department of fish and wildlife;
(iii) External peer review: Review by persons that are external to and selected by the department of fish and wildlife;
(iv) Open review: Documented open public review process that is not limited to invited organizations or individuals;
(v) Legal and policy document: Documents related to the legal framework for the significant agency action including but not limited to:
(A) Federal and state statutes;
(B) Court and hearings board decisions;
(C) Federal and state administrative rules and regulations; and
(D) Policy and regulatory documents adopted by local governments;
(vi) Data from primary research, monitoring activities, or other sources, but that has not been incorporated as part of documents reviewed under the processes described in (i), (ii), (iii), and (iv) of this subsection;
(vii) Records of the best professional judgment of department of fish and wildlife employees or other individuals; or
(viii) Other: Sources of information that do not fit into categories i - vii.

Meeting the Intent of RCW 34.05.271

Assigning references to categories requires judgement where methods of peer-review are not clearly defined. We assigned all scientific journal articles, science related books by independent science book publishers, universities, the National Research Council (NRC), and the National Academy of Engineering; symposia volumes sponsored by professional organizations (e.g., American Fisheries Society); and graduate theses and dissertations to category i –*independent peer review.* Government agency documents including those published by the Fish and Wildlife Service (USFWS), Environmental Protection Agency (EPA), Geologic Survey (USGS), Forest Service (USFS), and the Canadian Wildlife Service were assigned to category *viii*. Others references were assigned as described above.

Headwater wetland that flows north to Stavis Creek and south to Tahuya River/Keith Folkerts, WDFW

Washington
Department of
FISH and
WILDLIFE