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May 16, 2018

TO: Riparian Stakeholders and All Interested Members of the Public

FROM: Terra Rentz, Ecosystem Services Division Manager

SUBJECT: *Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications*

We are pleased to release our draft Priority Habitats and Species (PHS) guidance on riparian ecosystems in Washington. This two-volume set consists of a science synthesis (volume 1) and management recommendations (volume 2).

The attached document, *Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications*, has been reviewed, edited and re-reviewed by the Washington State Academy of Science (WSAS). This review process produced a document that met WSAS standards for a synthesis of current science across the range of topics we covered. At this time, Volume 1 is complete and is not open for public comment.

The department is seeking comments on *Riparian Ecosystems, Volume 2: Management Recommendations*, available at https://wdfw.wa.gov/publications/01988/ throughout a 60-day comment period, concluding on Tuesday, July 17, 2018. Volume 2 is an implementation manual for how to protect functions and values of riparian ecosystems and surrounding watersheds, using best available science synthesized in Volume 1.

To submit written comments on Volume 2, please visit
https://wdfw.wa.gov/conservation/phs/mgmt_recommendations/comments.html.

Together, Volume 1 and Volume 2 update and expand information provided in our 1997 PHS Riparian Management Recommendations. Thank you for your assistance making this a useful document.

Attachment: *Riparian Ecosystems, Volume 1: Science Synthesis and Management Implications*

RIPARIAN ECOSYSTEMS, VOLUME 1: SCIENCE SYNTHESIS AND MANAGEMENT IMPLICATIONS

A PRIORITY HABITATS AND SPECIES DOCUMENT OF THE WASHINGTON DEPARTMENT OF FISH AND WILDLIFE

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The scientific rigor of this document was improved through the peer review provided by the Washington State Academy of Sciences, with coordination provided by Dr. Robert Bates.

Draft versions of this document were reviewed by numerous staff in state agencies, including the Department of Ecology, Department of Natural Resources, Department of Commerce, Recreation and Conservation Office, State Conservation Commission, and Puget Sound Partnership. These reviews improved the document and we are appreciative of the helpful input we received from the state family.

We also appreciate the review by federal and tribal entities including EPA, NOAA Fisheries, Northwest Indian Fisheries Commission, Columbia River Inter-Tribal Fish Commission, Upper Columbia United Tribes, and individual tribes.

Finally, we acknowledge and express our appreciation for all local governments and individuals who provided feedback to us during our public comment period.

While we acknowledge and have deep appreciation for all the review and comments provided, WDFW bears sole responsibility for this document and any errors contained in it.

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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 protect fish and wildlife—public resources the agency is charged with managing and perpetuating. WDFW works cooperatively with land use decision makers and landowners to facilitate land use solutions that accommodate local needs *and* needs of fish and wildlife. WDFW's role 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 an earlier document entitled *Management Recommendations for Washington's Priority Habitats: Riparian* (Knutson and Naef, 1997). 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 later. Until the terrestrial species update is completed, 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. To qualify as a "Priority Habitat" in WDFW's PHS program a habitat must provide unique or significant value to many species. It must meet at least one of the following criteria (WDFW, 2008):

- Comparatively high fish and wildlife density
- Comparatively high fish and wildlife species diversity
- Important fish and wildlife breeding habitat
- Important fish and wildlife seasonal ranges
- Important fish and wildlife movement corridors
- Limited availability
- High vulnerability to habitat alteration
- Unique or dependent species

Riparian areas meet all of these criteria. Because of the many important ecosystem services (hydrologic, geomorphic, and biological) riparian areas provide, they were among the first PHS Priority Habitats identified and described by WDFW.

The PHS program provides land use decision support to clients such as local governments, developers, agencies, tribes, and landowners. PHS consists of PHS List, PHS Maps (available online at http://wdfw.wa.gov/mapping/phs/), PHS Management Recommendations, Technical Assistance (available from our Regional Habitat Biologists), Customer Service, and the newest component PHS Adaptive Management Support.

This PHS riparian document compliments a family of PHS document 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 http://wdfw.wa.gov/conservation/phs/mgmt_recommendations/)

LIST OF ACRONYMS

BAS	Best Available Science
BFW	Bankfull width
BMP	Best Management Practice
CAO	Critical Areas Ordinance
CMZ	Channel Migration Zone
DBH	Diameter at breast height
DNR	(Washington) Department of Natural Resources
DO	Dissolved oxygen
DOC	Dissolved organic carbon
EPA	Environmental Protection Agency
FEMAT	Forest Ecosystem Management Assessment Team
GIS	Geographic Information System
GMA	Growth Management Act
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
RMSE	Root-mean-square error
RMZ	Riparian Management Zone
SMA	Shoreline Management Act
SMP	Shoreline Master Program
SPTH	Site-Potential Tree Height
SPTH$_{200}$	Site-Potential Tree Height (200-year-old tallest dominant trees)
TAG	Technical Advisory Group
USFS	Unites State Forest Service

USFWS	United States Fish and Wildlife Service
USGS	Unites States Geological Survey
WAC	Washington Administrative Code
WDFW	Washington Department of Fish and Wildlife
WSAS	Washington State Academy of Sciences

GLOSSARY

Adaptive management: The systematic acquisition and application of reliable information to improve management over time. It treats management decisions as experiments in order to address critical *uncertainties* and learn more quickly from experience. It involves setting targets, monitoring benchmarks, and adjusting management decisions based on results. The hallmarks of a sound adaptive management program are: 1) adequate funding for research, 2) a willingness to change course when pre-established triggers are reached, and 3) a commitment to gather and evaluate conditions at appropriate spatial extents for necessary time scales. See Ecosystem-based management.

Anthropogenic: Related to human activity.

Aquatic species: Wildlife species that live in freshwater including fish, shellfish (clams, snails, mussels), amphibians (e.g., frogs, salamanders), turtles, crustaceans (e.g., crayfish), insects (e.g., larval mayflies, stoneflies, caddisflies, dragonflies) and various other invertebrates.

Bias (scientific): The phenomenon of gathering information that is not representative of the system as a whole. It can result from study design, conscious decisions (e.g., selecting sites for an experiment that hold some variables constant to study the effects of the variable of interest), or unconscious actions (e.g., assuming a theory is true or false without evidence).

Channel confinement: An indicator of how much a channel can move within its valley determine by the ratio of valley width (distance between toe of hillslopes on both sides of a stream) to active channel width. Typically, a segment is considered confined if the ratio is less than 2 and unconfined if greater than 4.

Channel migration zone: The area within which a river channel is likely to move over a period (e.g., 100 years).

Channel reach (stream): A specific portion 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%.

Complexity: The complicated state seen in dynamic environments that contain multiple components and *processes* that interact with one another in a complex web of interactions whose outcomes are often unpredictable. Complexity can be described with conceptual models; outcomes of well-understood complex phenomena can be partially predicted using computer models.

Composition: A term describing the all parts of an ecosystem that include both living (biotic) and nonliving (abiotic) elements. Ecosystem composition is an important consideration in conservation.

Disturbance regime: The frequency, magnitude, and duration of *disturbance* events.

Disturbance: A temporary change in environmental conditions (*composition*, *structure*, and *function*) within an ecosystem.

Dynamic equilibrium: An ecological system's long-term state of relative stability which is brought about through opposing, dynamic forces and continual states of flux. Activities such as urbanization, forestry, windthrow, landslides and forest fire can compromise dynamic equilibrium in riparian–stream systems. Protecting a *watershed's* dynamic behavior (rather than any specific feature) is an overarching goal of ecosystem-based *riparian* management.

Ecological integrity: The *structure*, *composition*, and *function* of an ecosystem operating within the bounds of natural or historical *disturbance regimes*. See Historical condition and Range of natural variability.

Ecosystem composition: All living (biotic) and nonliving parts of an ecosystem.

Ecosystem function(ing): 1) The *process* or the cause-effect-relationship underlying two or more interacting components, e.g., terrestrial plant material as food/substrate for aquatic invertebrates, 2) The sum of *processes* that sustain the system, and 3) the capacity of natural processes and components to provide goods and services that satisfy human needs, either directly or indirectly. Ecosystem functions can be conceived as a subset of ecological processes and ecosystem components and structure (see ecosystem process).

Ecosystem process (or ecological process): Complex interactions between biotic (living organisms) and abiotic (chemical and physical) components of ecosystems through the universal driving forces of matter and energy (see ecosystem functioning).

Ecosystem structure: The arrangement of and relations among the parts or elements (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: Management driven by explicit goals, executed by policies, protocols, and practices, and made adaptable by monitoring and research based on our best understanding of the ecological interactions and *processes* necessary to sustain ecosystem *composition*, *structure*, and *function*. EMB acknowledges that humans are an important ecosystem component and focuses on managing human activities within ecosystems. EMB often involves balancing ecological, economic, and social objectives 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 in *riparian areas* diminishes its ability 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 to stream as a function of distance from channel (measured in fraction of a *site-potential tree height*).

Flow regime (stream): The distribution of stream flow 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) and within a stream exert a strong influence on the flow regime.

Function: Discrete *ecosystem processes* used to define the ecosystem. See Ecosystem Function(ing) and Ecosystem process.

Historical condition: The dynamic state of a place prior to the arrival of non-indigenous peoples. It is the conditions under which native species evolved, and therefore, represents conditions that should most reliably maintain resilient self-sustaining native fish and wildlife populations. It is useful as a reference point (or conceptual model) for understanding how managed an area such that it moves in the direction of greater *ecological integrity*. See Ecological integrity and Range of natural variability.

Hot moments (*nutrient cycling*): Periods of elevated denitrification rates. Hot moments can occur during a rainfall event.

Hot spots (*nutrient cycling*): 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: The longitudinal, lateral, and horizontal movement and storage of water.

Hyporheic zone: The area beneath and alongside a stream channel where surface water *infiltrates* and exchanges with subsurface flow.

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 increase flows capable of transporting (scouring) sediment.

Infiltration: The rate or *process* by which water on the ground surface enters the soil.

Keystone species: A species whose ecological effect are disproportionate to their abundance and biomass, e.g., salmon and beaver.

Keystone ecosystem: An ecosystem whose effect on the broader ecosystem is disproportionate to their size, e.g., *riparian ecosystem*.

Keystone processes: *Ecological processes* that have widespread impacts throughout an ecosystem, e.g. *riparian* forest succession, *riparian* nutrient uptake, flood flows.

Macroinvertebrates (benthic): Animals, including insects, mollusks, crustaceans, and worms, that live within streams, do not have a backbone, and are large enough to be seen without a microscope. They are important components of the ecosystem and are commonly used as an indicator of habitat and *water quality*.

Mass wasting: The down slope movement of material due to gravity (rather than water, wind, or ice, for example).

Monitoring and adaptive management: See Adaptive management

Morphology (stream channel, aka fluvial geomorphology): A stream channel's shape and how it changes over time as a result of the interplay of *hydrology*, vegetation, sediment movement, and its position within the landscape. Channel morphology is influenced by the abundance and variation in sediment sources, the ability to transport sediment downstream, and interactions of sediment with *riparian* and instream vegetation.

Novel (or engineered) **solutions**: Solutions that are not found in nature. Examples: *vegetative filter strips* designed to capture excess nutrients; dams designed to provide flood control; engineered logjams designed to provide streambank channel roughness and complexity.

Novel conditions or ecosystems: Conditions (ecosystems) that are without historical precedent. Example: wetlands created in arid regions by leakage from irrigation canals, presence of manmade chemicals in the environment, and climate change due to anthropogenic greenhouse gas emissions.

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 quality, topography, groundwater level, and soil type. Excess nitrogen and phosphorus—used extensively in fertilizers—can create significant problems such as eutrophication, harmful algal blooms, fish kills, and contamination of drinking water supplies.

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. Washington forests typically have relatively tight N and P cycles, with low rates of inputs and outputs, but high rates of internal cycling.

Population viability (local): The likelihood that a population of a species will persist for some length of time.

Precautionary principle: Erring on the side of not harming resources when faced with *uncertainty*, especially for harm that is essentially irreversible. Utilizing a 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 Ecosystem process

Range of natural variability (or Historical range of natural variability): natural variability refers to two intertwined concepts: 1) that past conditions and processes provide context and guidance for managing ecological systems today, and 2) that disturbance-driven spatial and temporal variability is a vital attribute of nearly all ecological systems.

Recruitment (wood): 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, and tree mortality due to fire, ice storms, beavers, insects, or disease. Dominant factors include channel width, slope steepness, slope stability, forest *composition* and *structure*, and local wind patterns.

Riparian area: The area in alongside a stream or river.

Riparian corridor: See Riparian area.

Riparian ecosystem: The area alongside a river or stream that significantly influences exchanges of energy and matter with the aquatic ecosystem. It includes the active channel, the active floodplain and terraces, and portions of the adjacent uplands that contribute organic matter and energy to the active channel or floodplain. It is a zone of influence; a transitional ecotone between terrestrial and aquatic ecosystems that is distinguished by gradients in biophysical conditions, *ecological processes*, and biota.

Riparian habitat: see Riparian area

Riparian Management Zone: A delineable area defined in a land use regulation. RMZs are often used to protect *riparian ecosystems* and can be subdivided (e.g., core/inner/ outer RMZ) to provide varying levels of protection.

Riparian zone: See Riparian area

Riparian: An adjective meaning “alongside a stream or river.”

Risk: A situation involving exposure to danger, harm, or loss. Risk reflects the magnitude of the adverse impact and its probability of occurring. Risk is appropriately managed by applying *the precautionary principle* (especially for irreversible losses) and through *adaptive management*.

Riverscape: The landscape in which *riparian ecosystems* interact. It 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, 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 of which salmon and trout are members. Salmonids in Washington include Chinook salmon, chum salmon, coho salmon, pink salmon, sockeye salmon/kokanee, steelhead/rainbow trout, cutthroat trout, and bull trout/Dolly Varden.

Shifting baseline syndrome: A gradual lowering of standards or expectations for what constitutes a “degraded” ecosystem. The shifting baseline syndrome may be the result of each new generation perceiving what they experience as “normal” or “natural.”

Site class: The classification of a site based on the productivity of its dominant tree. 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 (200 years or more) for a given site class.

Stochastic event: An event which is randomly determined (e.g., landslide, flood). Stochastic events 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 the Tri-Cities the Columbia River is a 10th order river.

Structure: See Ecosystem 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 the state of something or a relevant variable. Uncertainty can be the result of natural variation (i.e., because outcomes vary in difficult-to-predict ways through time and space), model uncertainty (i.e., we do not understand how things interact with each other), systematic error (e.g., poorly designed experiments or calibrated instruments), or measurement error. Appropriate management responses to scientific uncertainty include gathering site-scale information, monitoring and *adaptive management*, applying the *precautionary principle*, and applying robust solutions (e.g., solutions that are likely to perform well over a range of conditions). See Risk.

Vegetative filter strips: *Novel solutions* designed to capture water transported nutrients, contaminants compounds and sediment.

Water quality (riparian): Physical, chemical, and biological characteristics of water that describe its suitability to meet human needs or habitat requirements for fish and wildlife. *Riparian areas* affect water quality by intercepting, accumulating and cycling fine sediments, excessive nutrients, and contaminants in overland and shallow subsurface flows.

Watershed processes: The fluxes of energy (e.g., sunlight, wildfire) and materials (particularly water and sediment) that interact with biota (e.g., vegetative cover, 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: A landmass that drains to a common waterbody.

CHAPTER 1. INTRODUCTION

Timothy Quinn, Kirk Krueger and George Wilhere

1.1 DOCUMENT DESCRIPTION

Volume 1 is the first of a two-volume set. It contains syntheses of scientific information for the purpose of informing the development of policies related to the management of riparian and watershed areas of Washington State. Volume 1 adds additional information to the science summarized in the 20-year-old PHS report entitled *Management Recommendations for Washington's Priority Habitats: Riparian* (Knutson and Naef 1997).

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

- How do riparian and watershed areas affect habitat condition of fish and wildlife in rivers and streams?
- How do human activities affect the capacity of riparian and watershed areas to provide habitat for fish and wildlife in rivers and streams?

Volume 1 is intended to: 1) be a source of best available science (BAS), especially as required for land use planning and 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)), and 2) provide a scientific foundation of management strategies for Volume 2 that protect, preserve, and perpetuate Washington's fish and wildlife consistent with WDFW's mandate (RCW 77.04.012).

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 (Ecology). However, because water quality is affected by land use, this issue is discussed in the context of riparian and watershed management. 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 considered part of riparian ecosystems, which are the focus of this document. Other specific wetland management guidance can be obtained from Ecology.

Volume 2 translates these scientific findings into guidance to local governments and others to protect and manage riverine systems via land use practices. 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 aquatic systems and their associated riparian areas.

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 (components) of something complex” (Oxford Dictionary 2015). We define ecosystem function(ing) to include the first and second definition of ecological function by Jax (2005): 1) the process, or the cause-effect-relationship underlying two or more interacting components (e.g., terrestrial plant material as food/substrate for aquatic invertebrates), and 2) the sum of processes that sustain the system, respectively. 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 ecosystems—and their dynamics over short and long periods. 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, structures and function of ecosystems represent the arrangement and organization of biotic and abiotic elements in space and time in addition to the processes between and among elements. Further, the composition, structures and functions can also vary with the scale of description. These concepts are closely related but not equally well studied nor understood. For example, we may be better able to characterize ecosystem composition and structures (Noss 1992, Hunter et al. 1988) than we can functions even if functions are in large part responsible for maintaining key observable components and structures of ecosystem. Myster (2001) argues that ecosystem structure can be defined as the parsimonious pattern of organization necessary for a function to operate, and that some ecosystem structure is meaningless when there is no linkage between a function and structure. Others argue that managing for functions (processes), as in process-based restoration (Beechie and Bolton 1999), is likely to be more successful than managing for structure since processes are more directly linked to land use activities (Reid 1998) and likely exert stronger influence on structure than the other way around (Odum 1953). Such scientific uncertainty suggests that we consider the entire ecosystem, i.e., composition, structure and functions, when providing management considerations for riparian areas. The following chapters synthesize our review of the relationship between composition, structure and functions of riparian ecosystems in the Pacific Northwest. While much is known about the arrangement and organization of biotic and abiotic elements in space and time in addition to the processes between and among elements, much remains unknown. We often describe desired future conditions of riparian ecosystems in terms of structure and composition rather than functions per se for two reasons: 1) structure and composition are more readily observable and thus easier to describe than functions, and 2) structure and composition are frequently used to describe management goals.

1.1.2 Riparian Ecosystems

WDFW has adopted an inclusive operational definition of riparian ecosystems (NRC 2002):

> *“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 ecosystems are sometimes referred to by different names, e.g., riparian areas, riparian zones, or riparian habitats, and riparian zone is also sometimes described as a distinct region within riparian ecosystems (Sedell et al. 1989, Naiman et al. 1992, Steiner 1994). Some definitions include the adjacent waters whereas others do not. According to Steiner et al. (1994), different definitions of riparian ecosystems have hindered state and federal agency effort to protect these systems. Definitions that are more recent now include common themes or concepts and identify similar sets of components, structures and functions, including distinct plant communities and soil types; disturbance regimes unique to fluvial systems; gradients in vegetation, hydrology, and soils that manifest an ecotone between terrestrial and aquatic environments; and essential interactions between terrestrial and aquatic systems. Riparian areas are the loci of terrestrial and aquatic interactions, however, the exact boundaries of riparian ecosystems can be difficult to describe because ecosystems are not uniquely identified entities (Karr 1996) and because the environmental heterogeneity of riparian areas is expressed in a variety of plant life history strategies and successional patterns, while functional attributes depend on community composition and environmental setting (Naiman et al 1998). Unless otherwise specified, we consider riparian corridor, riparian habitat, riparian area and riparian ecosystem synonyms herein. Following Gregory et al. (1991) and Naiman et al. (1998) we define stream riparian ecosystems to include, the active floodplain including riverine wetlands, and terraces, and the adjacent uplands that directly contribute organic matter or large wood to the active channel or floodplain. Hillslope areas that provide wood and sediment to streams via landslides are included in this definition. See Figure 1.1.

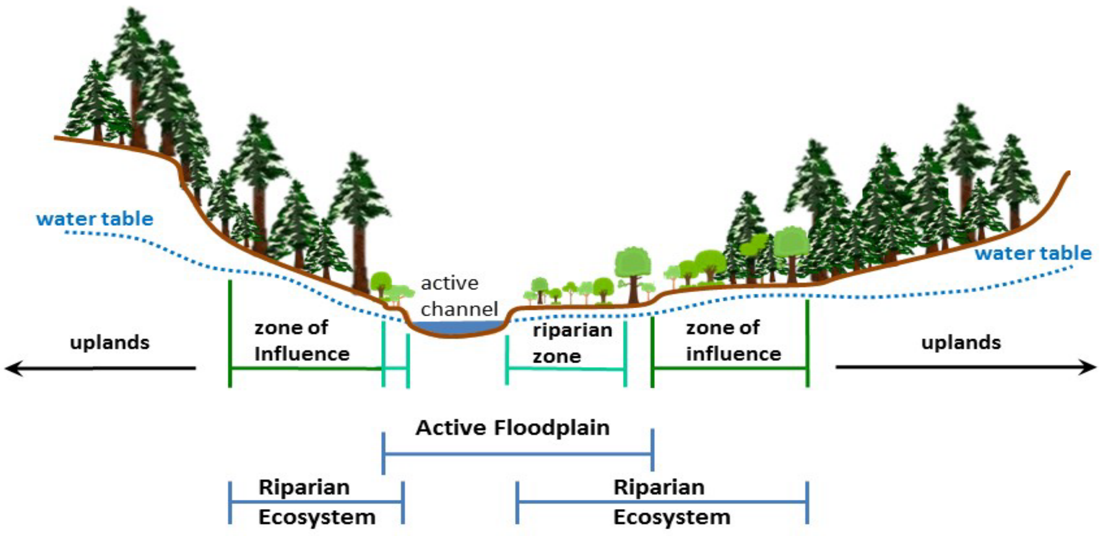


Figure 1.1. A generalized diagram of the riparian ecosystem. The NRC (2002) states that, "Riparian areas are transitional between terrestrial and aquatic ecosystems and are distinguished by gradients in biophysical conditions, ecological processes, and biota. They are areas through which surface and 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)." We describe NRC's (2002) riparian areas as *riparian ecosystems*, the portion of the ecosystem distinguished by gradients as *riparian zones*, and the area of the terrestrial ecosystem that influences the exchange of energy and matter as the *zones of influence.* The width of the zone of influence is typically based on site-potential tree height measured from edge of active channel or floodplain.

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 (Thomas et al. 2006). 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 complete their life cycle (T. Quinn, unpublished data). 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 ecosystems are characterized by the availability of water, mild 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 of 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 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 wdfw.wa.gov/publications/00693/wdfw00693.pdf.

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 & Fish Agreement (1999). The riparian ecosystem functions we discuss in this document, including some associated compositional and structural elements, are organized by chapter:

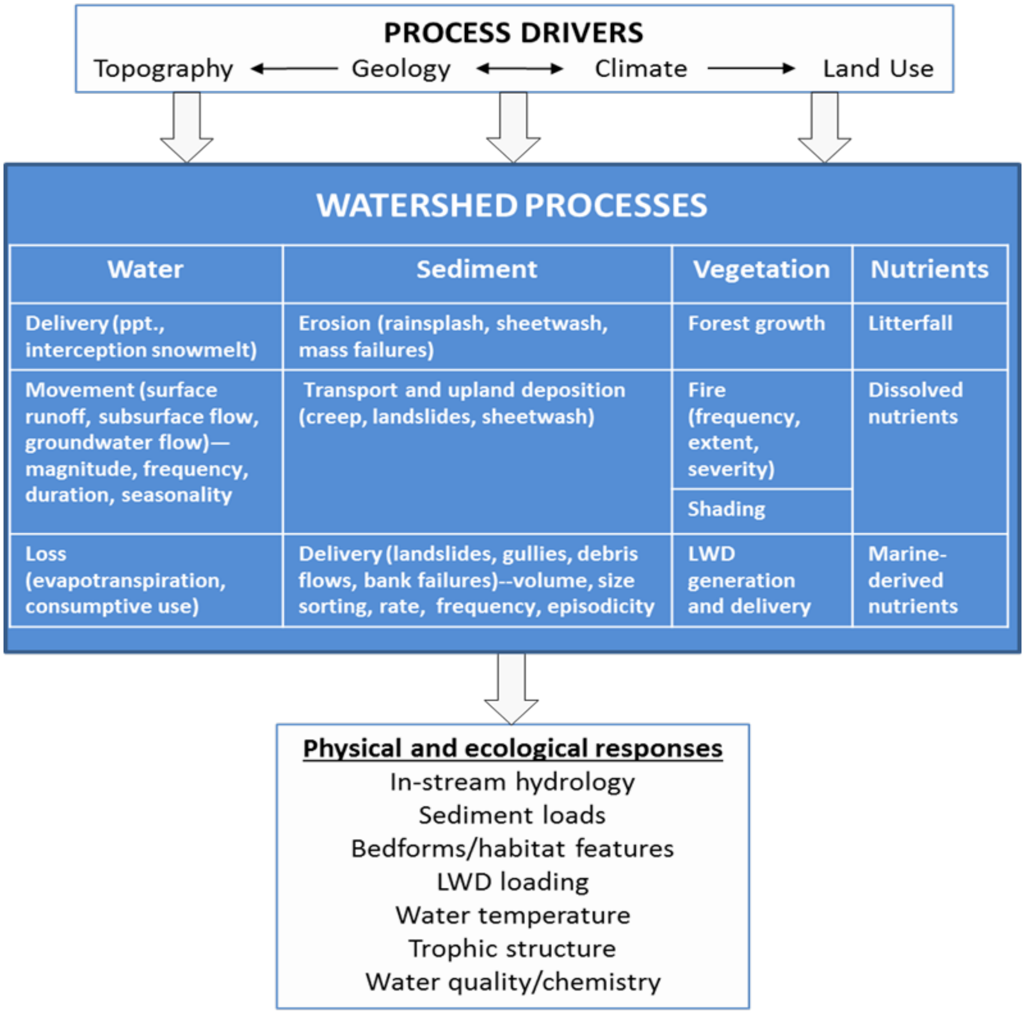
- Chapter 2. Hydrology and Physical Processes—the interplay of water, sediment and vegetation on channel form and riparian areas
- Chapter 3. Wood—effects of large wood on channel morphology and habitat, and the recruitment of large wood from riparian areas
- Chapter 4. Stream Temperature—effects of 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-nutrient: 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 (NRC 2002, Naiman and Bilby 1998). Second, local governments and individual landowners, important users of WDFW's PHS program 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 of 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 production, routing and storage of sediment, wood and fine organic matter; and the quality of water, thereby influencing riparian ecosystem functions (NRC 2002; Figure 1.2). Thus, the management of riparian ecosystem at the site or reach scale should be considered just one important part, but not the sole requirement, of holistic watershed management for the protection of fish and wildlife and other human values that must also include watershed-scale considerations. 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.



Water	Sediment	Vegetation	Nutrients
Delivery (ppt., interception snowmelt)	Erosion (rainsplash, sheetwash, mass failures)	Forest growth	Litterfall
Movement (surface runoff, subsurface flow, groundwater flow)—magnitude, frequency, duration, seasonality	Transport and upland deposition (creep, landslides, sheetwash)	Fire (frequency, extent, severity) Shading	Dissolved nutrients
Loss (evapotranspiration, consumptive use)	Delivery (landslides, gullies, debris flows, bank failures)--volume, size sorting, rate, frequency, episodicity	LWD generation and delivery	Marine-derived nutrients

Figure 1.2. 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."

1.2.3 *Science Synthesis*

Chapter 9, From Science Synthesis to Management Recommendations, 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, 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, the initial conditions, planning, management practices, regulatory, and legal context for protection are different from those for restoration, and thoroughly covering both topics was beyond our capacity. 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 a variety of approaches to developing this document. We first established a Technical Advisory Group (TAG) of applied scientists from the region to help provide project direction and to review draft written products. Second, with guidance from the TAG, we conducted a literature review designed to inform the general state of knowledge regarding ecosystem protection and restoration 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. To be clear, the chapters in this document are not literature reviews or technical updates of Management Recommendations for Washington's Priority Habitats: Riparian (Knutson and Naef 1997); they provide a more comprehensive description of the state of science on these functions that can be afforded by a focused literature 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. Importantly, given the broad potential applicability of this document, it is conceivable that all 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 riparian functions including hydrology, stream channel morphology, large wood supply, water quality, nutrients and water temperatures and watershed processes. Note that while many of the issues we address here are directly or indirectly related to fish and wildlife habitats and other ecosystem services, we did not 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, what we know, how well we know it and what we do not know. The summary statements in each chapter are especially important to the development of management recommendations because they help us avoid bias (e.g., preferential reference to only studies that support a conjecture) and 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 summarized and generalized to novel conditions or unstudied locations. In other words, to consider the body of literature statistically with regard to our ability to make precise and reliable predictions of the effects of specific management actions at specific locations.

Where information existed and as time allowed, we also attempted to summarize how riparian composition, structures and functions might 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 for deriving useful land use advice. We considered four general categories of land use: 1) Urbanization, which includes commercial, industrial, urban, suburban and rural development; 2) Agriculture, which includes cultivated crops and grazing uses, 3) Forestry, which includes timber management and associated activities such as road building and maintenance, and 4) Others 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 four computerized databases. We mostly focused our literature search to 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 biophysical conditions.

All studies that met initial search criteria (i.e., ~11,000 sources) were imported into EndNote® version X2 either as full text or abstracts. We checked these studies against the original four criteria (above) and then did more focused keyword searches of titles and abstracts in EndNote during which we could readily tag studies as *relevant, irrelevant* and *uncertain relevance*. We focused on retaining studies pertaining to: 1) riparian systems along freshwater lakes and rivers; 2) riparian functions as described above; 3) fish, benthic invertebrates, amphibians, reptiles closely associated with lotic systems; and 4) floodplain or riparian system wetlands. Just over 6,000 studies remained after we completed this review. Studies classified as *uncertain relevance* were read and then retained as *relevant* or removed from further consideration. Relevant papers were amassed into a project research library and assigned three additional key words that described a specific riparian function type, land use type, and species type where applicable. 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. We recorded in EndNote 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 the EndNote database as a useful source of information for writing the synthesis.

We found a massive quantity of literature related to riparian and watershed science despite the literature search was focused on recent information defined as being 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. 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 contractors for completing a chapter. The steps involved in completion of the drafts included adding one or more conceptual models describing the system and its functioning and a synthesis or summary of the important finding 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 work 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.

We enlisted the Washington State Academy of Science (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 contractors and TAG members, addressed WSAS comments, make 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 (ideally consensus) about scientific contents and conclusions.

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CHAPTER 2. STREAM CHANNEL MORPHOLOGY

Jane Atha, Mike Liquori, Kirk Krueger, and George Wilhere

2.1 INTRODUCTION

The composition, structure, and functions of streams and river systems are due to dynamic interactions of water, sediment, vegetation, wood, and other organic matter. The nature of these interactions is largely determined by topography (e.g., slope), sources and characteristics of sediment, the capacity of water to transport sediment, and the size, strength, and density of riparian vegetation (Naiman and Bilby 1998). The important influences of water and channel slope on sediment transport results in channel patterns (i.e., straight, meandering, and braided; Leopold and Wolman 1957) and geomorphic zones (i.e., source, transport, depositional; Schumm 1977) along elevation and slope gradients. Furthermore, differences in fluvial processes (e.g., frequency and intensity of floods and debris flows) and controls (e.g., lithology, riparian vegetation. and instream large wood) 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), disturbances (Frissell et al. 1986), and dams (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 affect it may be impacted by human activities (e.g., embankments, dams, and stream crossings), usually resulting in loss of habitat (e.g., stream area), less habitat diversity (e.g., pools, side channels), and reduced habitat functions (e.g., nutrient contribution, shading) for aquatic species. Management actions such as diking, riparian vegetation removal, and human development, tend to reduce natural variability of geomorphic processes, often amounting to stream habitat degradation greater than the sum of its parts.

This chapter provides an overview of water and sediment in fluvial processes, and their interactions with riparian vegetation, and potential alterations of fluvial processes by human activities. We first describe a conceptual model for considering how physical processes affect channel morphology. Next, we describe in more detail stream flow and sediment processes. We then consider the watershed-scale context that directly influences fluvial processes. We direct the reader to Chapter 8 for a thorough examination of influences and management that occur at the watershed scale. Lastly, we review the impacts of human activities on the function of streams and provide guidance on how to incorporate process domains and spatiotemporal scale into management. Some related topics are discussed in other chapters. For example, see Chapter 3 for a description of fluvial process interactions with large wood.

2.2 CONCEPTUAL CONTEXT

Knowledge of the physical processes that affect riparian and aquatic habitats is important for making sound policy and management decisions. The most widely used fluvial conceptual model posits that rivers in a healthy watershed exhibit “dynamic equilibrium” (Gilbert 1877). Dynamic equilibrium describes a state of relative stability that results from the interaction of opposing forces. Over a given time period, a channel’s composition and structure, such as the relative abundance of channel form types, remains largely unchanged. A stream may undergo significant changes after an extreme flood event, and yet return to dynamic equilibrium as smaller flows rework the sediment and form erosional and depositional features. Lane (1955) conceptualized equilibrium within a channel segment as a balance among stream discharge, channel gradient, sediment load, and sediment size (Figure 2.1). 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 causes 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. Within limits that are specific to geomorphic domains or channel forms, shifts in stream discharge, channel gradient, sediment load, or sediment size are accommodated through adjustments in one or more of the other variables, and consequently, the current state of dynamic equilibrium is maintained. However, disturbances of sufficient intensity, frequency, or duration can exceed a threshold that causes a shift to a new equilibrium state (Cluer and Thorne 2012).

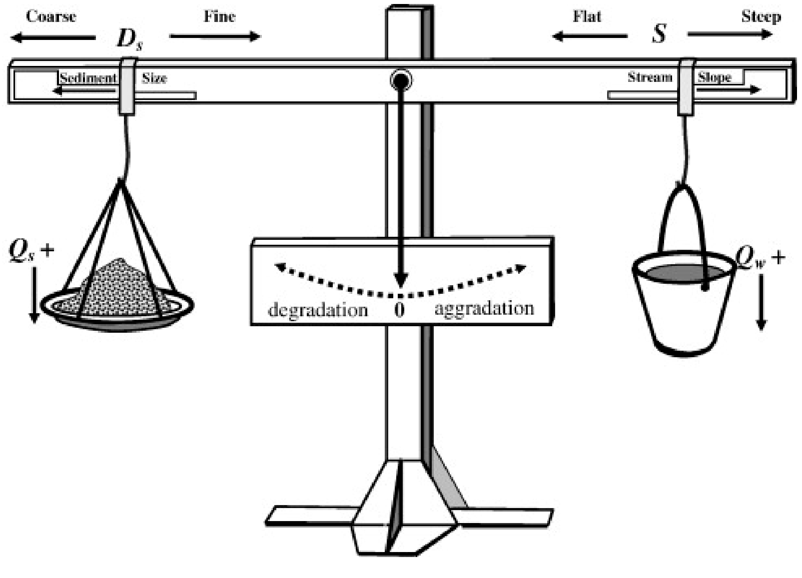


Figure 2.1. Lane’s Balance illustrating the concept of channel adjustment (i.e., vertical aggradation of degradation of the streambed) in response to changes in water or sediment yield (from Dust and Wohl 2012).

While Lane's Balance is a useful conceptual model, it omits an important element affecting fluvial processes. Channel morphology is also affected by interactions between sediments and riparian vegetation (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 shape riparian plant communities. Human activities that alter these interactions can cause significant changes to channel morphology and fish habitats.

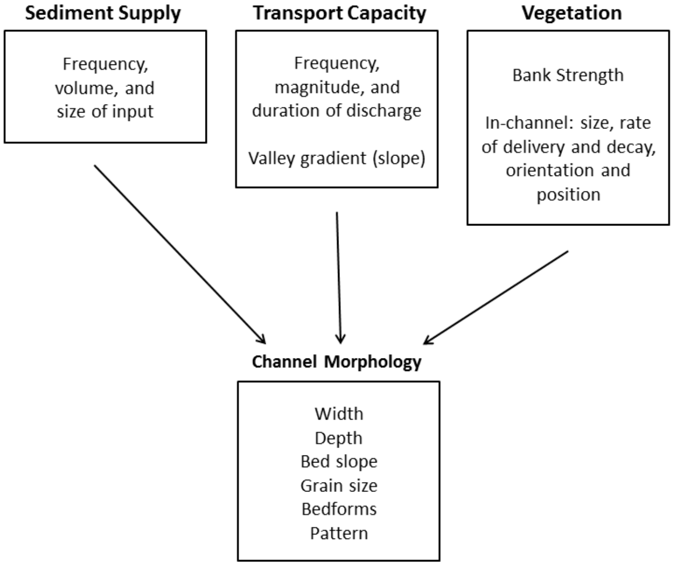
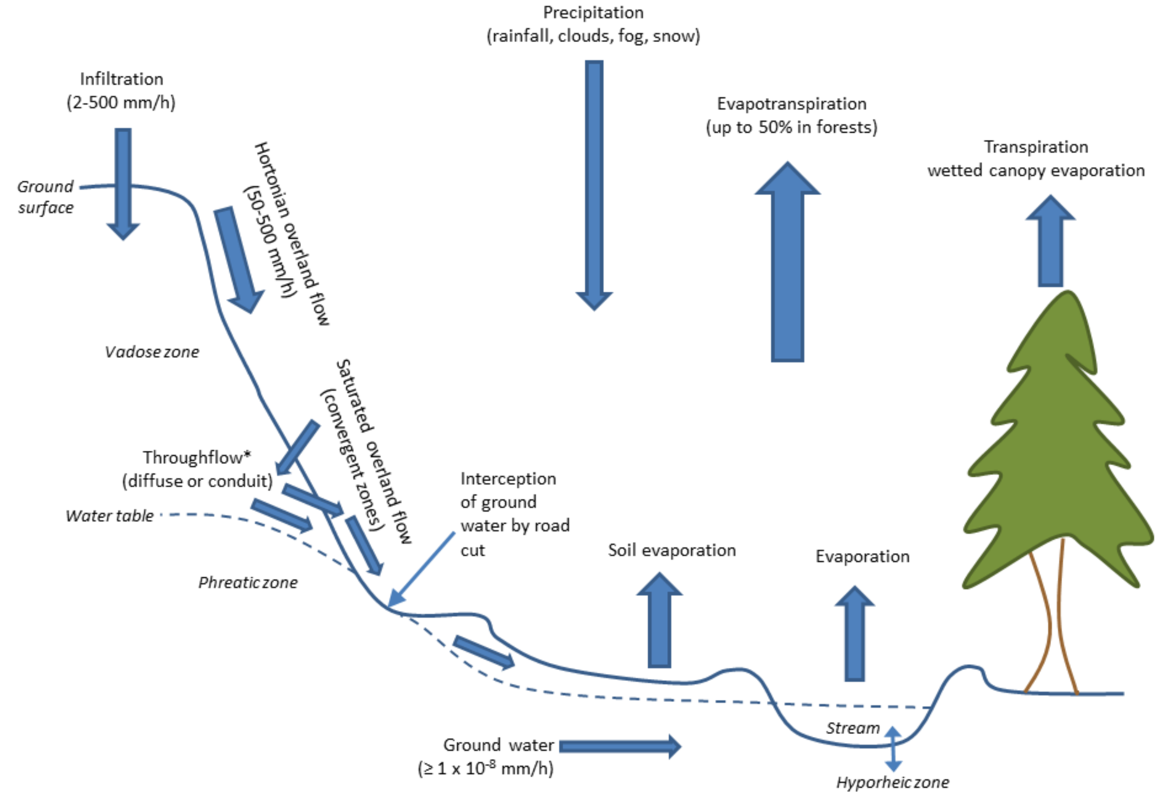


Figure 2.2. Decade-to-century scale influences on channel morphology (Naiman and Bilby 1998).

Many geomorphic and ecosystem processes occur in parallel to and at similar spatial and temporal scales resulting in a mutual dependence (Renschler et al. 2007). While these parallel couplings have received much attention within the last fifteen years in the scientific community, a theoretical synthesis with a revision of conceptual frameworks is still needed to address how riparian management interacts with these complex systems (Smith et al. 2002; Stallins 2006). Interdisciplinary research of these system interactions could result in better predictability of impacts of management actions across scales. From an ecosystem perspective, the choice of spatiotemporal boundaries—often shaped by geomorphic processes—is of profound importance to conceptualization of the system and the ecologic processes under investigation (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. This conceptual basis of river system adjustment (*sensu* Lane 1955) is important to consider when examining each process independently.

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 temporal variability in environmental conditions produces a wide range of stream flow responses to precipitation events. The fate of precipitation that is not returned to the atmosphere by evaporation and transpiration is highly dependent on soil cover. Water that infiltrates into the subsurface 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).



* Pipe flow 50-500 mm/h; diffuse flow 0.005-0.3 mm/h

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).

When precipitation exceeds infiltration capacity (Horton 1933) or when the local soils are saturated (Dunne and Black 1970a; Dunne and Black 1970b; Dunne 1978), water moves downslope as surface runoff. Surface runoff remains outside the confines of a channel, and is most common where vegetation is sparse. Human activities that compact soil or cause loss of permeable, near-surface soil layers (the most extreme being impermeable urban surfaces), promote overland flow, a relatively rare occurrence in natural watersheds. 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).

The spatial distribution of 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 thresholds of activation for lateral subsurface flow that occurs in the period immediately following precipitation (McDonnell 2003).

Riparian vegetation influences the manner in which surface flows create and maintain the channel and floodplain morphology, as well as preferential flow pathways in both surface and subsurface environments (Swanson et al. 1998; McDonnell 2003). During periods of active runoff, riparian areas quickly become saturated, and are the first watershed domain to begin contributing 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. The flow resistance provided by riparian vegetation and related organic debris slows water velocities, reduces peak discharge, and lengthens a flood's duration (Tabacchi et al. 2000; Nilsson and Svedmark 2002). Alternatively, hydrology plays a large role in shaping riparian vegetation communities. The rates of flood stage influence 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).

Water infiltrates and exchanges with a subsurface flow beneath the stream channel called the *hyporheic zone*. The flow paths of the hyporheic zone originate at the channel and can extend more than 2 km laterally from the channel in wide valleys and to depths of 10 m (33 ft; Stanford and Ward 1988). It underlies the riparian ecosystem and is important in regulating water quality (Tabacchi et al. 2000; Chapter 5). Two-way exchanges of water between the stream and the riparian aquifer are important in regulating several ecosystem processes (Moore and Wondzell 2005). As part of hyporheic exchange, water, sediment, solutes, and small organisms such as microbes and macroinvertebrates can strongly influence the volume, temperature, and chemistry of streamflow (Chapters 4-6). The hyporheic zone can account for a fifth of the invertebrate production in an aquatic ecosystem (Smock et al. 1992). Dense riparian vegetation locally limits infiltration into the hyporheic zone, and contributes to sediment trapping during floods. This process contributes to the modification of the moisture-related properties of the substrate (Tabacchi et al. 2000). Hyporheic flows can also affect riparian vegetation, although the interactions between riparian communities and hyporheic conditions are not well understood (NRC 2002).

2.4 EROSION AND SEDIMENTATION PROCESSES

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 weather patterns, availability of materials, and changes in exposed surface area.

2.4.1 *Sediment Sources and Delivery*

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Sediment can move downslope in aggregates through mass movement categorized as landslides, debris flows, or heaves. Mass movements 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 strength of the slope materials (soils, roots, etc.). Mass movements are initiated when the shear stress on the material exceeds its shear strength threshold (Ritter et al. 2002). The volume of sediment generated from hillslope mass wasting events can be very large, and is often the predominant source of sediment delivered to streams in the Pacific Northwest (Swanson et al. 1982).

Sediment can 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 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 and unable to be smoothed by ordinary farm tillage), 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 are associated with treefall that result in the uprooting of root balls (root throw; Hairston-Strang and Adams 2000; Liquori 2006). The primary causes of treefall include wind along newly created riparian vegetation edges (Liquori 2006; Reid and Hilton 1998), undercutting streambank erosion, and 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). Windthrow-related sediment production from riparian areas can be responsible for delivering 21 to 32 tons of sediment per mile of stream length (Gomi et al. 2005).

Riparian vegetation can effectively trap and store sediment transported downslope from overland flow (Naiman et al. 2010). 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). Many riparian species are able to grow deeper roots as they are buried in new pulses of hillslope sediment.

2.4.2 *Instream Sediment Transport and Deposition*

Sediment entering 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. The slower rates of movement allow for longer reaction times with the surrounding environment, and 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, such that the amount in transport is limited by the amount available for transport. By contrast, the coarser bed load consists of larger rocks and debris that bounce along the channel bed, and are typically limited by transport capacity of the flow water (stream energy). Bed load may not get suspended; it can roll and slide during low flows, saltate (leap) at intermediate flows, and move as sheetflow during high flows (Wohl 2014). The bed load transport process is surprisingly complicated because there are many factors, some of which interact, that determine the threshold for the movement of sediment, such as the range of sediment sizes, their spatial distribution 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 stream 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. 2006).

All bedforms and associated classifications above 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 energy and 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 and 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, a headcut can occur (Wells et al. 2009). Headcuts that occur at the channel head result 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; Foster 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.

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 the interactions among sediment, water, 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 how small plant-scale phenomena that structure patch-scale geomorphological forms and processes affect interactions between patches. These interactions are crucial to understanding longer-term and larger scale geomorphic form and process (Gurnell 2014).

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 which promotes the ecological diversity of riparian areas, and it 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. Bank erosion can also be caused by instream large wood or mid-channel gravel bars that divert flows toward a bank (Florshiem et al. 2008). 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. However, human disturbances can cause excess bank erosion, and 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 which 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 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. However, 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 (ASCE 1998, Simon and Collison 2002) as well. The weight of vegetation adds a "surcharge"[1] load to soil. Hence, the weight of vegetation increases the vertical shear stress near a streambank and adds to the lateral earth pressure[2] 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's stabilizing hydrological effect.

The bank stability function of riparian vegetation has important implications for riparian area management. FEMAT (1993, p. V-27) created a generalized curve of root strength versus distance from channel. It was based on expert opinion informed by the scientific literature. The radius of Douglas-fir root networks are well-correlated with tree crown radius (Smith 1964)[3] 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 diameter breast height) have root network radii around 18 to 34 ft Hence, in an undisturbed old-growth riparian

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[1] 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.

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

[3] The ratio of root spread to crown width for small Douglas-fir and western hemlock (mean dbh about 13 inches) was reported by Smith (1964) to be about 1. Data in Eis (1987) for these same tree species suggest a ratio of about 2.

forest, the full contribution of root strength to streambank stability is provided by trees within approximately 35 ft of the streambank.

2.4.4 *Lateral Channel Migration*

Lateral channel migration, operating like a conveyor belt, carves diverse, complex, and dynamic, aquatic and riparian habitats across a river's floodplain (Naiman et al. 1993). Lateral channel migration affects local sediment erosion and deposition, local topographic relief, flood inundation patterns, river planform, alluvial architecture, and riparian vegetation patterns. 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).

Meandering 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 migrating or occupying the valley floor (Dunne and Leopold 1978). The timeframe needed for a channel to migrate and occupy its entire floodplain can be calculated as a "floodplain turnover rate" (O'Connor et al. 2003), which may take hundreds to thousands of years in western Washington.

At meander bends the sediment erosion and deposition patterns that result in channel migration are caused by complex flow. There is an inherent tendency for water flowing over a rough surface 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 point bar deposition. In addition, point bars deflect flow laterally toward the cut bank, creating topographically driven secondary flow that further induces an outward velocity shift toward the cutbank (Güneralp and Marston 2012).

Migration of individual meanders can occur through four different processes: 1) translation, which is a downstream shift without significant shape change; 2) extension, where the amplitude of the bend extends by migrating across the valley; 3) rotation, in which the bend axis changes orientation and compound growth; and 4) lobing and compound growth, where the bends become less regular and symmetrical (Daniel 1971). 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 exceed a stability threshold causing a chute cutoff that creates a shorter channel across the inside of a point bar 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 can persist for decades to centuries depending on rates of sediment filling, and increase habitat diversity for aquatic and riparian organisms (Wohl 2014).

Avulsions are the sudden abandonment of a part or the whole of a meander belt by a stream for some new course, often 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 accumulating in and plugging channels, diverting flow into a relict channel, and making it the main channel (Collins and Montgomery 2002). This process also creates side channels, which are known to be important rearing habitat for juvenile salmonids (Rosenfeld et al. 2008), and oxbow lakes which are habitat for amphibians (Henning and Schirato 2006).

Lateral channel migration initiates spatial and temporal patterns in vegetation structure and composition that are controlled by differential rates of channel movement and vegetation response (Figure 2.4). As channels migrate laterally, point bars create unique recruitment sites for primary vegetation colonization. Over long periods, directional point bar growth and extension is followed by the succession of terrestrial plant communities. Consequently, a point bar may support a chronological series of successional stages arranged from youngest to oldest along a spatial transect (Shankman 1993). A limited number of species consistently dominate the accreting edge of pointbar landforms, while progressively more species establish as landforms aggrade 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 species or regenerating the existing community. 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 can leave human communities at risk along river systems. Likewise, human alterations to limit channel migration and erosion can degrade aquatic and riparian habitats. For these reasons, geomorphologists have developed protocols for determining the channel migration zone (CMZ). A channel migration zone includes the outer extent of known historical channels, plus potential future migration over the next 100 years, and they typically encompass floodplains and some portion of terraces (landform remnants of the former floodplain). CMZ delineation considers the historical migration zone, which is the collective area that the channel has migrated through in the historical record, the avulsion hazard zone, or areas not in the historical record that are at risk of avulsion, and also the erosion hazard area, which is the area at risk of bank erosion from stream flow or mass wasting over the timeline of the CMZ (Rapp and Abbe 2003). The CMZ also includes channels and terrace banks that are at risk of mass wasting due to erosion of the toe.

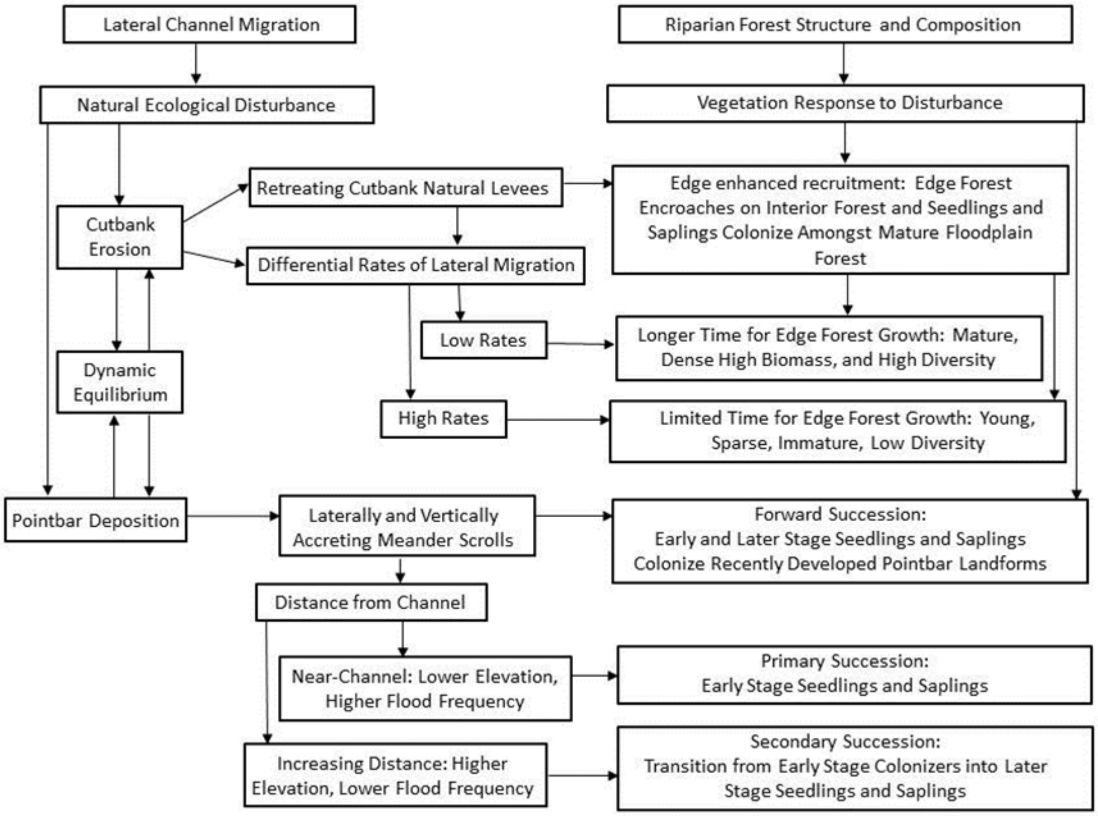

Figure 2.4. A conceptual model of lateral channel migration processes and riparian forest responses to cutbank erosion and pointbar development (modified from Meitzen 2009).

2.5 WATERSHED CONTEXT

Watershed conditions influence aquatic habitat conditions because various spatial domains of a watershed affect differently the processes and disturbances that manifest riparian functions (Nakamura and Swanson 2003). 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). Disturbances, and where they occur within process domains of a watershed, play key roles in structuring the riparian ecosystem. 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 encompass linkages between watershed-scale conditions and natural disturbance regimes (Naiman and Bilby 1998). The following section describes the latest understanding of geomorphic process domains and natural disturbance regimes that continuously shape the drainage.

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 channel forms. 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 six common geomorphic process domains within the forested watersheds typical to 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, including streams that initiate in perched wetlands.

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 flow 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 these domains, it is possible to identify watershed areas dominated by different geomorphic processes, disturbance regimes, response potential, and recovery time, as well as predict riparian vegetation widths (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. In higher gradient channels, energy is expended on transporting sediment downstream and is not available to reshape the riparian area.

Technical advances in mapping and geospatial data, such the availability of Geographic Information Systems (GIS) and remotely sensed data over the last several decades, have refined our descriptions and understanding of the processes that shape stream systems at a range of spatial and temporal scales. These tools aid in the extrapolation of parameters such as gradient ranging from a channel reach to a region. For example, using base maps in GIS to identify site locations, followed by field visits, Polvi et al. (2011) developed an effective model correlating riparian width with valley width and process domain classification in upland watersheds. Both parameters can be measured remotely in many cases. In order to completely consider and distinguish process domains within a watershed, it is also important to have knowledge of its disturbance regime—and how it may vary within a watershed—to best delineate and manage riparian width at the reach scale.

2.5.2 *Natural Disturbance Regimes*

Natural disturbances that occur within and among geomorphic process domains result in dynamic habitats of varying physical conditions that allow for a variety of species to coexist. Disturbances that influence streams and riparian areas include wildfire, flood, mass movements, infestation, disease outbreaks, and human-induced disturbances such as development, timber harvest, and land use conversion, among others. Community succession theory recognizes that large-scale disturbances, such as low frequency, high magnitude floods or mass movement events, 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 riparian areas.

Disturbance processes are essential for developing the range of habitat conditions necessary to support native biota (Swanson et al. 1998; Tabacchi et al. 2000). A natural disturbance regime can be characterized by frequency (how often), magnitude (how big), duration (how long), and predictability. For example, fires or large floods of a given magnitude 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 disturbance such as fire (Wright and Bailey 1992), insect (Mattson and Addy 1975), and disease-induced mortality (Matson and Boone 1984) 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 system could manifest a new ecosystem state with different physical conditions that support a different biological community (Hughes 1997).

Human activities can either dampen or amplify disturbance processes. While the disturbance processes are ongoing, they are supplemented by human alterations (e.g., dams eliminate or dampen flood flows and land management can reduce the frequency and severity of fires). If there is too little or too much disturbance, it 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 and disconnection to the floodplain thereby altering chemical and ecological processes. Conversely, too much flooding due to land cover change can reduce water storage and increase runoff leading to less complexity and fewer habitat elements within the channel and floodplain. 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.

2.6 HUMAN ALTERATION IMPACTS

Thus far, this chapter has largely focused on the hydrologic and physical processes and their interactions with riparian area vegetation as they naturally occur within a watershed. However, human influence on channel function and form is ubiquitous, and therefore should be considered in order to best inform potential management practices in riparian areas.

How a channel responds to human alterations of the landscape and riparian areas specifically depends on its sensitivity or resilience (Brunsden and Thornes 1979). A sensitive channel reach will likely respond to a change in external forces while a resilient channel reach will likely return to initial conditions following alterations. The idea that some parts of a river are more sensitive or resilient is critical to predicting channel adjustments. For example, stream segments with high transport capacity may adjust relatively quickly to an increase in upstream sediment yields whereas a similar change in sediment yield may cause significant changes in channel morphology where segments are transport limited. Channel geometry, hydraulics, and natural disturbance regime of a channel may be used to determine the sensitivity of a reach 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, hydraulic sensitivity is inversely proportional to channel gradient (Wohl et al. 2007). Hence, managers should anticipate that a high gradient stream would be more sensitive to changes in stream flow. Other drivers such as the flow regime or controls such as lithology might also be used to characterize stream process at finer spatial scales to guide management strategies for different human activities.

2.6.1 *Forestry*

An extensive body of scientific literature documents the effects of forestry on riverscapes globally (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 within a decade, and increases water yields over multiple decades as vegetation recovers (Wohl 2014). Impacts from road networks—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). It is a challenge, however, for researchers to distinguish channel changes that occur annually via natural processes of erosion and deposition from long-term trends in channel adjustments caused by human disturbances. 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.

There is substantial agreement among scientists as to the importance of the magnitude and consequence 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), hydrologic domain, harvest treatment, and channel type. The authors generally noted that geomorphic effects 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 review studies (e.g., Liquori et al. 2008; Grant et al. 1999) had similar conclusions.

The degree to which water and sediment yields are altered by forestry practices depends on where the logging occurs within a drainage basin. In one study, logging at lower elevations in snowmelt-dominated basins in 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, site 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 500 mm (22 inches) per year average precipitation compared with 50% of the sediment when precipitation exceeds 3,000 mm per year (118 inches; Coe 2006). In areas of high relief, it is a particular challenge for researchers to differentiate channel adjustments induced by logging from change that would have occurred without interference (Marston 2008).

Deforestation, and associated land use conversion, of riparian areas causes 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, and simpler, with lower habitat diversity, fewer obstructions, higher proportions of run and glide habitat, and less variability in active channel width (Jackson et al. 2014).

2.6.2 *Road Infrastructure*

Road networks are similar to stream networks in that they are often widely distributed, are “designed” to 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). Road construction and maintenance disturbs a layer of soil on the road tread, ditch, and cutslope that becomes the most 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, roads, trails, and ruts are the most significant management activity that affects sediment production and delivery into streams (Lewis et al. 1998; Gomi et al. 2005; CBOF TAC 2008; others). 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 (native versus rocked), road slope, mean annual precipitation, geology, traffic type and volumes, and road area (WA DNR 1997; Coe 2006; MacDonald et al. 2004; Cafferata and Munn 2002; others). 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 (Cafferata and Munn

2002; Madej 2001). Jones et al. (2000) found that unpaved roads continue to contribute sediment fines to channels long after harvested vegetation has reestablished.

2.6.3 *Agriculture*

Agriculture, in the form of livestock grazing and crop production, can reduce and alter vegetation cover, compact soils, reduce water infiltration; increase runoff and sediment yield, and cause associated changes in streamflow, channel morphology, and channel stability. It is well understood that vegetated riparian buffers are a primary defense against these impacts to water quality. A *buffer* is a strip of land managed to protect a resource or resources from adverse impacts. A riparian buffer is managed to protect aquatic resources from the impacts of upland activities.

Cattle grazing can generate severe degradation of 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 and adequate pool area, wide and shallow channels, high sediment yields and turbidity, steep gradients, and channel beds typified by long stretches of glides or runs. Magilligan and McDowell (1997) found that in areas that have enclosures keeping cattle out of riparian areas, channel widths became narrower due to increases in 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).

The most 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 soil production and natural erosion by up to several orders of magnitude (Montgomery 2007).

An important role of riparian buffers in agricultural areas is to intercept sediments flowing overland from upslope land disturbances and block pollutants in the form of excessive sediment that often contain contaminants (Asmussen et al. 1977). Additionally, riparian vegetation in agricultural areas can slow surface runoff and increase soil infiltration of water, thereby enhancing both deposition of solids and filtration of water-borne pollutants. Riparian buffers also intercept and act on contaminants in subsurface flow, and reduce sedimentation and associated nonpoint source pollution in these areas. Sweeney and Newbold (2014) provide a review of the efficacy of vegetated areas to protect excessive sediments from reaching river and streams. They concluded from their meta-analysis that removing ~65% of sediments from overland flow requires a buffer about 33 ft (10 m) wide and removing ~85% of sediments requires a buffer about 100 ft (30 m) wide. We direct the reader to Chapter 5 for details on the impacts of excessive sediment to water quality and aquatic habitat.

2.6.4 *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 deliver as much as 80% of the sediment yield from an urbanizing basin (Fusillo et al. 1977). Thus, active construction sites should be the focus of remediation efforts. Sediment yield per unit area tends to decrease with increasing drainage area because of "dilution effects", that is, as the proportions of areas under construction to watershed area decreases (Wolman and Schick 1967). For example, sediment production in Issaquah Creek in western Washington has two- to five-fold increases when only 0.3% of its 144 km^2 (55.6 sq. mi.) drainage area is under construction (Nelson and Booth 2002). Increased sediment yield can result in channel aggradation and planform changes. Upon completion of construction, sediment decreases to negligible amounts as urbanizing areas are stabilized beneath roads, buildings, and lawns. The next sequence of events in the urbanization process commonly includes changes to a stream channel, i.e., bed coarsening, incision, and bank erosion.

Research on the effects of urbanization on hydrology and flooding is also extensive, and date 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).

2.6.5 *Stream Channel Modifications*

Humans have been directly altering channel flow and form for centuries (Poff et al. 1997). Flow regulation, through dams, reduces the mean and coefficient of variation of annual peak flow (Williams and Wolman 1984), increases minimum flows (Hirsch et al. 1990), shifts the seasonal flow variability, and can greatly increase diurnal flow fluctuations if the dam is used for hydroelectric power generation (Magilligan and Nislow 2001). Numerous methods have been developed to quantify hydrologic changes due to dams, including the most common indices of hydrologic variation to determine the range of variability.

The impacts of dams on riparian communities are great in number (Nilsson et al. 1997; Nilsson and Berggren 2000; Katz et al. 2005; Merritt and Wohl 2006). Lateral disconnection of the channel and floodplain as a result of reduced peak flows from dams decreases or eliminates freshly scoured surfaces necessary for seedling establishment. Flow regulation changes can disrupt the downstream transport of seeds to germination sites by streamflow, and seedlings can also be killed from prolonged submersion due to increased base flows. Changes in sediment size distributions and moisture content can also limit riparian establishment. The cumulative effects on riparian communities downstream of dams can result in fewer plant species and decreased vegetation cover (Jansson et al. 2000).

The alterations humans make to channel form and connectivity are diverse in form and intent. Channel 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 a form more esthetically pleasing are all ways that humans regularly alter channel form (Wohl 2014). Levees are a direct form of channel alteration particularly ubiquitous in lowland rivers. Levees are linear mounds constructed along stream channels to limit overbank flooding and are also known as dikes, dykes, or embankments among other things (Petroski 2006). Levees severely reduce or eliminate channel-floodplain exchanges, facilitate higher magnitude floods, and exacerbate flooding in downstream areas without levees. Disconnections with the floodplain result in a reduction of sediment organic matter, and organisms in the channel, leading to loss of habitat, animal abundance, and biodiversity in riparian and channel environments (Hohensinner et al. 2004).

Lastly, humans can indirectly and severely alter streams and channel form by introducing invasive, exotic riparian species. Invasive species can outcompete 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 streamflow and riparian water tables, as well as nutrient cycling and the quality and quantity of riparian habitat for other species (Schilling and Kiniry 2007; Hultine and Bush 2011).

2.7 FISH AND WILDLIFE HABITAT

2.7.1 *Fish*

The fisheries and fish ecology literature are replete with studies that identify correlations between physical habitat conditions and fish populations (e.g., species occurrence, abundance, and density). Such studies demonstrate the profound importance of instream conditions for fish and they have helped guide management and restoration. For example, Nickelson et al. (1992) found seasonal differences in the use of pool and riffle habitat by juvenile coho salmon 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 configurations and optimal habitat ratios (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) which 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).

We suggest that the scientific advances over the last few decades that are most pertinent to the management of riparian areas to protect and restore fish habitat include the following elements: 1) acknowledgement of the importance and frequency of fish movement (Gowan et al. 1994; Schlosser 1991; 1995), 2) greater acknowledgement of the importance of habitat heterogeneity and spatial variability in fish-habitat relationships (Torgersen et al. 2006), and 3) the associated 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) largely by explicitly incorporating spatial and temporal heterogeneity, discontinuities, and connectivity between stream reaches (longitudinal), the stream and uplands (lateral), and the stream and groundwater (horizontal). 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 (components and structures) can be very important, 4) unintended consequences of habitat degradation occur in all directions, including upstream, and 5) studies should be conducted at the scale at which management can affect change. 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, we suggest that 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 addressing many questions requires study at large spatial and long temporal extents and monitoring several biotic and abiotic processes at fine spatial grains and temporal frequencies. 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 at a range of temporal frequencies with the intent of identifying rare features and measuring changes in habitat complexity and connectivity (Bennett et al. 2016). This approach will provide reliable inference to the efficacy or restoration efforts and can be used to inform decisions via adaptive management (Bennett et al. 2016).

2.7.2 *Amphibians and Reptiles*

Amphibians are less obvious components of instream habitats than fish, but in much of the Pacific Northwest, key groups of stream-breeding amphibian species (giant salamanders [*Dicamptodon* spp.], tailed frogs [*Ascaphus* spp.], and torrent salamanders [*Rhyacotriton* spp.]) dominate fishless headwater systems, which represents a large majority (up to 80%) of the overall stream network based on its length (Meyer and Wallace 2001). Stream-breeding amphibian dominance in such landscapes 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, clean, coarse refuge space, 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 do not attach their eggs, deposit them concealed 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, both of 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 outside of the headwater landscape at least in part because substrate and flow conditions that provide either oviposition or refuge sites become less frequent (Brummer and Montgomery 2003). This does not mean to imply that other conditions, such as predation by fishes, does not also play a role.

The freshets (high water events) in larger streams or rivers that have some kind of an alluvial floodplain, when coupled to an input of large wood, are also critical to both creation and maintenance of off-channel 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; Jones et al. 2005) and the garter snakes that feed on them; such off-channel habitats are also critical rearing and foraging habitat for native turtles (Holland 1994).

2.8 CONCLUSIONS

Streamflow and sediment dynamics coupled with their interactions with riparian vegetation create complex channel morphologies and diverse aquatic habitat conditions. Reach-scale differences in sediment delivery, transport, and deposition result in a mosaic of aquatic habitat conditions within and among stream reaches. While the concepts of dynamic equilibrium and geomorphic process domains are useful, heterogeneity of habitat conditions among stream reaches should be expected and considered in management decisions, especially when discontinuities (e.g., tributary junctions or culverts) are present. It is often possible to delineate process domains at a site with knowledge of ecological drivers—such as gradient and natural disturbance regime—as well as management history.

Natural and anthropogenic disturbances affect the creation, maintenance, destruction, and recreation of aquatic habitats within the habitat mosaic. As previously discussed, disturbances and the resulting bio-geomorphic interactions are increasingly acknowledged as vital to maintaining habitat conditions required by aquatic species. Thus, effects of human actions on the type, location, frequency, intensity, duration, and predictability of disturbances should be a critical management consideration. However, it is important to consider that habitat conditions are often limited by a site's (and its watershed's) historical and current management that alter disturbance regimes and related processes that affect the channel and corresponding habitat.

Management may most reliably result in high quality habitat when it emulates historical disturbances and their effects on a watershed (Poff et al. 1997). One approach put forward by Naiman et al. (2000) is to base management on probability distributions of historical patterns of watershed conditions and natural disturbances. With empirical data or models on the extent to which flood disturbances, for example, have affected channel and floodplain characteristics, one can estimate probability distributions of channel and riparian floodplain conditions over space and time. Consequently, riparian management actions could be based on assessments of the disturbance frequency of a watershed and be tailored to the unique characteristics of individual watersheds. For example, if fires and landslides are key disturbances to a watershed, reserves may be identified that preserve essential riparian function but logging can be allowed in other locations in the watershed (Naiman et al. 2000).

Disturbance and successional processes are necessary for the maintenance of stream and riparian ecosystem composition, structure and function. Therefore, understanding of a channel's state of equilibrium (or disequilibrium) and corresponding trajectory is critical to appropriate riparian management. Not all locations within a channel network are equally sensitive or resilient to land conversion or other anthropogenic disturbance. An understanding of the historical range of natural variability of channel process can facilitate evaluation of a site's potential response to human disturbance. This perspective may also aid in identifying unique biological communities that are in most need of riparian protection (McDonald et al. 2004; Brierley and Fryirs 2009). Using a strategic (rather than opportunistic) landscape management perspective, we may be able to move closer to historical processes and functions and the resulting habitat heterogeneity essential for watershed health. The composition, structure, and disturbance history of natural reference sites may be useful for understanding the historical range of natural variability.

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 bank stability are increasingly emphasized. Management of riparian ecosystems should consider spatiotemporal scales of stream function, connectivity among the geomorphic domains, the influences of natural disturbance, the historical range of natural variability at a site, and the inextricable influence that riparian vegetation has on geomorphic processes.

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CHAPTER 3. WOOD

George 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: 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). Consequently, 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, however, some important questions remained unanswered. The main purpose of this document is to provide a scientific foundation for managing the terrestrial portion of riparian habitats. This literature review and synthesis will focus on: 1) the role of wood in aquatic ecosystems, and 2) the recruitment of large wood to aquatic ecosystems. This chapter does not cover stream restoration involving the artificial placement of 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). Riparian forests are an important source of instream large wood, but this review does not cover riparian forest management. 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 a critical role in the composition, structure and function of riparian and aquatic ecosystems. In forested ecoregions, wood is the primary determinant of channel form and dynamics in small streams (Montgomery and Buffington 1997, Bilby and Bisson 1998), however, the magnitude of wood's effects is affected by channel dimensions and 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). These many effects of large wood create a variety of channel forms—dam pools, plunge pools, riffles, glides, undercut banks, and side channels—which provide a diversity of aquatic habitat types. 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, and therefore, are important food production sites for juvenile salmonids (Bisson et al. 1987). Furthermore, pools provide rearing habitats and essential refuge from high flows for fish and other aquatic fauna. The structural presence of large wood provides fish with cover from predators, and by increasing a waterbody's effective space, wood structures may increase fish densities (Bisson et al. 1987).

3.2.1 *Small Wood*

Wood is typically divided into two size categories: large and small. Large wood is usually defined as greater than 10 cm (4 in.) in diameter and greater than 2 m (6.5 ft) in length (Bilby and Ward 1991, Schuett-Hames et al. 1999)[1]. Small wood consists of branches and other woody material not classified as large wood.

Small wood plays essential and unique roles in lotic ecosystems. For instance, accumulation of small wood enhances a stream's retention of leaves and other particulate organic matter (Gregory et al. 1991), which are vital food sources for many aquatic invertebrates. Small wood also effects channel geomorphology. Plunge and dammed pools are often associated with accumulations of small wood, and large wood with dense accumulation 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 beaver dams (Naiman et al. 1988). Beaver dams store large quantities of sediment, reduce channel incision, remove excess nutrients from water, increase water retention and base flows, reduce peak flows and spread flows out 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 North American beaver (*Castor canadensis*) is lacking. However, beaver most often forage on trees ranging from 3-8 cm (1.2-3.2 in.) 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 10 cm (4 in.) 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 interaction on the role of beaver and beaver dams in aquatic ecosystems, see Pollock et al. (2015).

3.2.2 *The Role of Instream Large Wood*

The main role of large wood in aquatic ecosystems is large roughness element (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 habitat types.

The influences of instream large wood on aquatic ecosystems are a function of wood size relative to channel width. 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

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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 common definition is greater than 10 cm (4 in.) in diameter and greater than 1 m (3 ft) in length. Small wood is also known as small woody debris or fine woody debris.

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 control 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 promote stable channels (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 (e.g., wood, boulders) in small non-fish-bearing streams in 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, lower stream power mobilizes less sediment, less sediment movement results in increased sediment storage, and more sediment storage leads to greater channel stability.

Table 3.1. Minimum dimensions of large wood key pieces for different channel widths. Piece diameter is measured at midpoint of piece length.

Bankfull Width (m)	Minimum Volume (m^3)	Example Dimensions	
		Length (m)	Diameter (m)
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 175 m (575 ft) of streambed. Beschta (1979) observed a similar effect after woody debris dams were removed from a moderate gradient headwater stream in western Oregon—5,000 m^3 (6,500 yd^3) of sediment was lost from 250 m (820 ft) of streambed over a period of about 10 months.

Large wood also causes 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 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 channels. These disturbances to channel geomorphology create a shifting mosaic of aquatic and riparian habitat types.

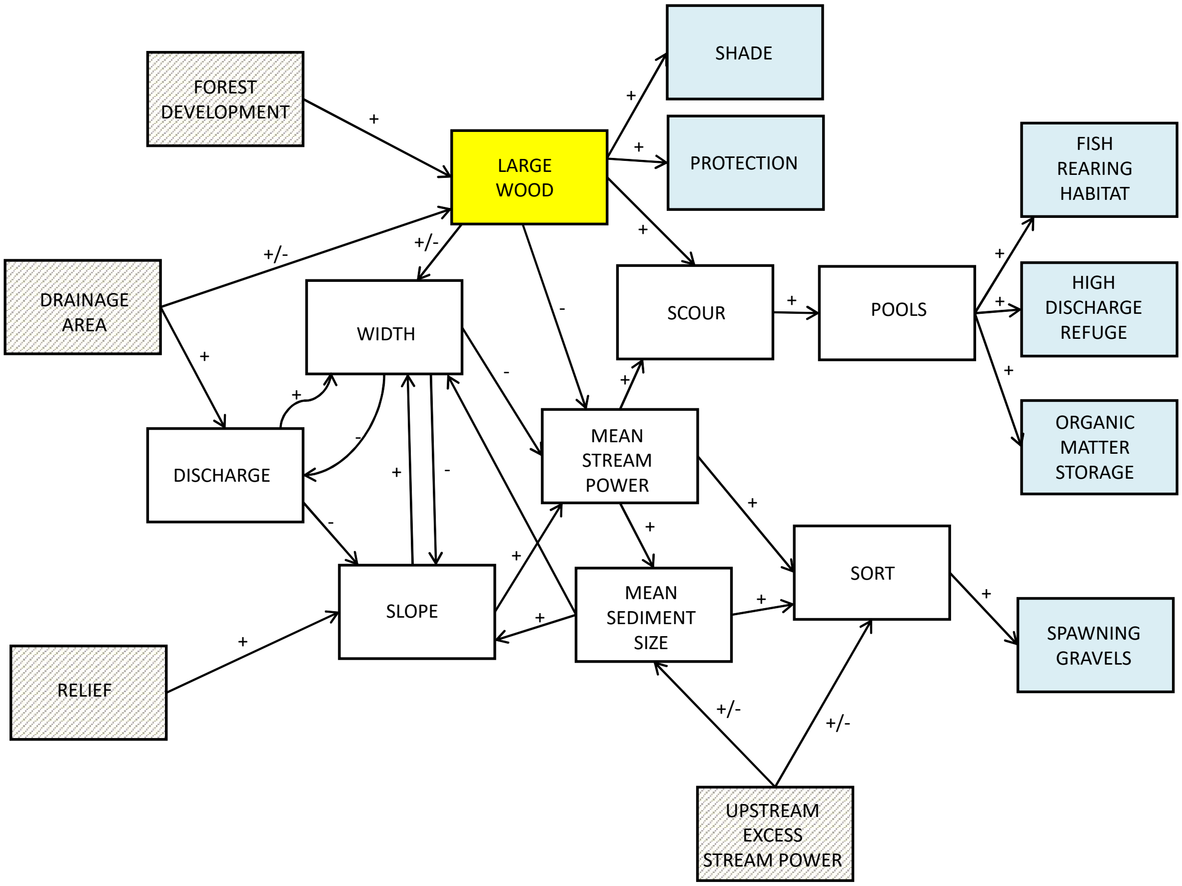


Figure 3.1. Conceptual process-response model of large wood and sediment in fluvial systems. Arrows indicate the hypothesized directions of causal relationships; pluses and minuses indicate the hypothesized slope of each relationship. The hatched variables are exogenous to the system, and the blue variables are the associated habitat features. The white variables are endogenous and constitute the stream channel sub-system (modified from Rhoads 1988) with the addition of large wood.

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 debris, 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. Large wood altering channel morphology by creating dam pool, step drop, and plunge pool.

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 directly 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 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 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 water storage. For example, 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 debris dams held 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 the time-course of high flow events by reducing peak discharge and increasing the event's duration (Gurnell et al. 1995), however pools created by debris dams raise the water table in adjacent streambanks (Gurnell et al. 1995). This is likely to affect the composition riparian vegetation. For instance, at some sites, wet riparian soils may be maintained through seepage from pools, and wet soils are more suitable for alder (*Alnus rubra*) and cedar (*Thuja plicata*, *Chamaecyparis nootkatensis*) than other tree species. Alder is known to contribute enormous amounts of beneficial nitrogen to soils and streams (Naiman et al. 2002b).

Figure 3.3. Sediment and nutrient storage and gravel bar formation behind large wood.

The hyporheic zone is the saturated sediment beneath a stream channel and under 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 reduce a streams thermal sensitivity (Chapter 4).

3.3 RECRUITMENT OF INSTREAM LARGE WOOD

The quantity and quality of instream large wood are determined by the processes of wood recruitment, decomposition, and transport (Figure 3.4). The term “recruitment” is often used to describe the process of wood moving from the terrestrial environment to the stream channel. The terms “transport,” “import,” and “export” are used to describe wood moving within the channel; wood is imported to a stream reach and exported from a stream reach.

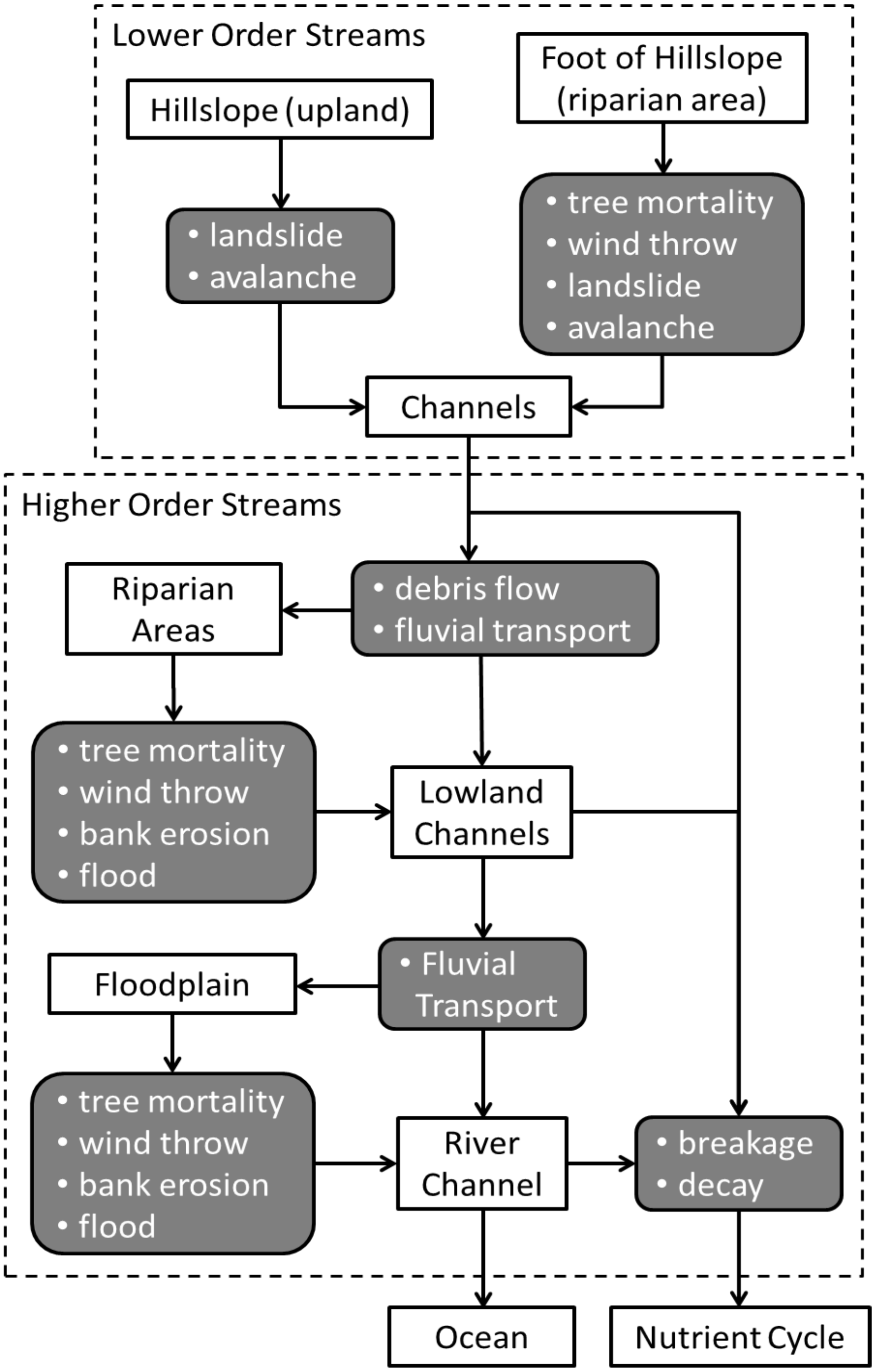
Decomposition and transport occur within the stream channel, and hence, are outside the document’s scope which focuses on the terrestrial portion of riparian habitats. For our purposes, we will focus on the recruitment process, which occurs in riparian forests. For comprehensive, in-depth reviews of scientific literature pertaining to the dynamics of instream large wood, including decomposition and transport, see Benda et al. (2003), Bilby (2003), Gurnell (2003), and Piégay (2003).

3.3.1 *Wood Recruitment Mechanisms*

Recruitment of large wood to aquatic ecosystems occurs by disturbances such as bank erosion, windthrow, landslides, debris flows, snow avalanches, and tree mortality due to fire, ice storms, insects, or disease (Swanson et al. 1976, Maser et al. 1988). At any given site, all mechanisms may contribute to the recruitment process, however, one mechanism may dominate, and the dominant mechanism is determined by channel width, 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 smaller, confined streams can recruit instream large wood via several mechanisms; however, landslides and windthrow are the primary mechanisms of large wood delivery 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 suppression, insects or disease because smaller streams recruit less wood through lateral bank avulsion. Beavers 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 was the primary driver for recruiting large wood in alluvial channels, while windthrow largely drove large wood recruitment in bedrock channels (Murphy and Koski 1989). Over a 63 year period, the Queets River (bankfull width = 128 m, 420 ft), recruited 95% of large pieces (≥ 1 m diameter) through erosion that occurred as its unconfined channel meandered across its wide floodplain (Latterell and Naiman 2007).

The relative contribution of each mechanism to instream wood 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%.



2250 **Figure 3.4. Wood recruitment mechanisms and wood transport pathways (adapted from Hassan et al. 2005).**
2251 **Lower order streams are first and second order headwaters.**

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.

		Murphy and Koski (1989)	**Johnston et al. (2011)**	**May & Gresswell (2003)**	**Benda et al. (2003)**
	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
	location	southeast Alaska	south & central British Columbia	Coast Range Oregon	northern California
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

A site's recruitment mechanisms determine the magnitude and time-scale 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 to streams 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 Coastal Range (May and Gresswell 2003). In this study, 80% of wood pieces and total wood volume originated from forests within 50 m of colluvial channels constrained by steep hillslopes, whereas in unconfined alluvial channels, 80% of instream large wood originated from within 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 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 90 m upslope of the channel (Reeves et al. 2003). More specifically, 65% of instream pieces of large wood and 46% of wood volume originated from these upslope locations.

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	2nd to 5th	> 30 m (100 ft)	bank erosion	99% of pieces from < 30 m (100 ft). 45% of pieces from within 1 m (3 ft) of streambank. Average tree height = 40 m (130 ft) ‡.	1
Western Oregon & Washington	1st to 3rd	55 m (180 ft)	not reported	50% of pieces from within 10 m (33 ft) of streambank; 85% from within 30 m (100 ft); 90% from within 39 m (128 ft; horizontal distance). Ave. tree height = 57.6 m (189 ft).	2
Northern California	14-17 m (45-55 ft)	55 m (180 ft)	variable	90% of wood from < 30 m (100 ft; slope distance). Longer recruitment distances due to landsliding.	3
Coast Range Oregon	2nd and 3rd	70<d<80 m (230, 262 ft)	windthrow	In alluvial stream 80% of pieces from ≤30m (100 ft). In colluvial streams ≈50% of pieces from ≤30m (100 ft). For both types ≈75% of volume from ≤50 m (165 ft).	4
Coast Range Oregon	4th	>90 m (300 ft)	landslide & debris flow	About 65% of pieces and 46% of wood volume were from upslope sources by landslides or debris flows.	5
Olympic Peninsula	6	>450 m (1,475 ft)	channel migration	95% of wood from < 265 m (870 ft). 50% of pieces from within 92 m (300 ft).	6
Southeast Alaska	5-30 m	35< d <40 (115, 130 ft)	variable	96% of all large wood from within 20 m (65 ft) and 89% of from within 10 m (33 ft). Average tree height = 22.6 m (74 ft)	7
South & Central British Columbia	1-17 m (3-55 ft)	65 m (425 ft)	tree mortality	90% of large wood pieces and volume delivered to stream channels from within 10 and 9 m (33 and 30 ft), respectively. Average tree height= 37 m (120 ft).	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. The source distances for approximately 90% of instream large wood from mature and old-growth conifer riparian forest in western Washington and Oregon were within 26 m and 40 m (85 and 130 ft) 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 = 57.6 m, 189 ft) compared to trees in mature riparian forest (mean height = 48 m, 157 ft). 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 asymptotic (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 30 m of the streambank, but 45% of large wood originated from within 1 m (3 ft, Murphy and Koski 1989). Likewise, in a study from Oregon and Washington, 100% of large wood with an identified source was recruited from within 55 m (180 ft), but 50% of large wood with an identified source originated from 10 m (33 ft, 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.

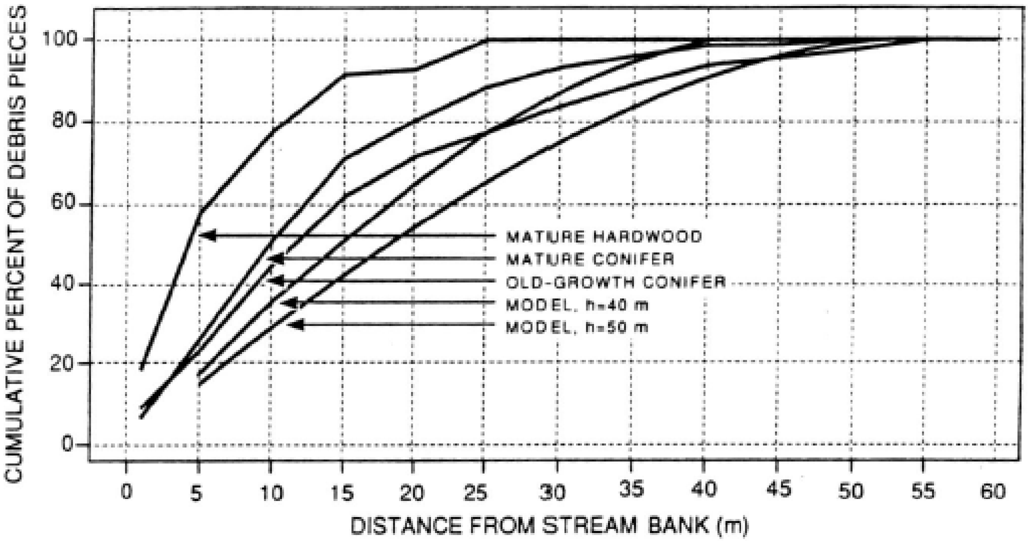


Figure 3.5. Distribution of source distances from tree origin to streambank. Empirically derived curves for old-growth conifer, mature conifer, and mature hardwood stands in western Oregon and Washington. Modelled source distances for trees 40 m and 50m tall (from McDade et al. 1990). Empirically derived curves based on large wood with an identified source.

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 90 m from the stream channel (Reeves et al. 2003). In 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 92 m (300 ft) 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 impacts 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 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 only 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 buffers and much higher tree densities in riparian areas; regulations should result in a stand basal area equal of 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 consequently, the composition, structure, and functions of riparian areas have yet to attain a dynamic equilibrium.

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 adequate fish habitats is unknown, and resolving that issue will be difficult until riparian areas approach their 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, “natural” riparian forests. 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 landsliding 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 30 m, but due to landsliding two sites had large wood recruited from beyond that distance—32 m (105 ft) and 65 m (210 ft).

The results of Benda et al. (2002) and Benda and 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 landsliding dominates 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 structure, 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 one 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 feet in diameter were pulled from a 50-mile 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[2] 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 extremely limited at present. Nevertheless, we believe 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 grasslands. 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

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[2] Forest zones that are largely determined by regional climatic differences (Franklin and Dyrness 1988). Forest zones contain similar climax plant communities that are influenced by similar disturbance regimes. There are seven such forest zones in Washington.

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, WDFW, pers. comm.; B. Renfrow, WDFW, pers. comm.). 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 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 without excessive pressure from large herbivores (Mack and Thompson 1982).

With the exception of black cottonwood, woody plants in grassland and shrub-steppe regions contribute little large wood to stream channels; that is, only a small proportion of the stem is larger than 10 cm (4 in.) in diameter. Therefore, 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 led to lowering of water tables led to loss of riparian vegetation led to 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 develop reference conditions for instream wood and reference conditions 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 are needed to establish management objectives. Much work needs to be done on historical conditions of riparian areas in non-forested regions.

Table 3.4. Distributions of large wood per 100 m (330 ft) of stream channel by forest regions in Washington State and by bankfull width class (Fox and Bolton 2007). Large wood was defined as pieces exceeding 10 cm (4 in.) in diameter and 2 m (6.5 ft) in length. Key piece sizes defined in Table 3.1.

Region*	BFW Class (m)	75th Percentile	Median	25th Percentile
Number of Pieces per 100m				
Western Washington	0-6	>38	29	<26
	>6-30	>63	52	<29
	>30-100	>208	106	<57
Alpine	0-3	>28	22	<15
	>3-30	>56	35	<25
	>30-50	>63	34	<22
Douglas-fir & Ponderosa Pine	0-6	>29	15	<5
	>6-30	>35	17	<5
Number of Key Pieces per 100 m				
Western Washington	0-30	>11	6	<4
	>30-100	>4	1.3	<1
Alpine	0-15	>4	2	<0.5
	>15-50	>1	.3	<0.5
Douglas-fir & Ponderosa Pine	0-30	>2	0.4	<0.5
Volume (m^3) per 100 m				
Western Washington	0-30	>99	51	<28
	>30-100	>317	93	<44
Alpine	0-3	>10	8	<3
	>3-50	>30	18	<11
Douglas-fir & Ponderosa Pine	0-30	>15	7	<2

* Sitka spruce, western hemlock, silver fir, and mountain hemlock forest zones were grouped to form the "Western Washington Region". Subalpine fir and the grand fir forest zones were grouped to form the "Alpine Region."

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 cover only those taxonomic groups having the strongest associations with instream wood: fish, amphibians, reptiles, and invertebrates.

3.5.1 *Fish*

For salmonids, especially juvenile coho, 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, juvenile coho salmon, steelhead, and cutthroat trout in reaches with gradients less than 4% to greater than 10% (Bryant et al. 2007). Adult spawning sockeye 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). Coho salmon spawners 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 significant. Large pieces of instream wood provide fish with refugia 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 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, foraging habitat, and likely overwintering habitat 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 stream profiles 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 habitat (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 was positively correlated with the amount of functional large wood ($r^2 = 0.50$), and torrent salamander 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 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 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 larger, alluvial streams.

3.5.3 *Aquatic Invertebrates*

The distribution and abundance of all aquatic invertebrates is influenced by instream wood and certain taxa are dependent on instream wood. In streams of all sizes, areas where wood accumulations are present 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 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.

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. Aquatic insects are prey of juvenile salmonids. 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 consequently, 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/or size of trees in riparian areas. Furthermore, in association with all land uses, large wood is often purposefully removed from rivers or larger streams because it can: 1) block culverts or damage other human-made structures, such as bridges, dams, or levees; 2) cause streambank erosion that destroys private property; and

3) present hazards to navigation or recreation. Culverts and dams also cause "flow fragmentation" which disrupts the transport of wood through a stream network and reduces the amount of wood in higher order streams and rivers.

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 conifer trees in the Pacific Northwest have large diameter, decay resistant boles, streams in this region have high natural 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: 1) significantly higher in streams adjacent to unmanaged, mature and old-growth stands compared to streams along 30- to 50-year-old plantations, and 2) 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., ≤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 clear-cutting without a riparian buffer. They also estimated a 250-year recovery time to pre-harvest levels.

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, large wood generated from stand-replacing disturbances was depleted within 100 years of the event but once the forests re-established, 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 un-logged tracts. Consequently, wood recruitment increased by an average of 2 trees per 100 m (330 ft) in the buffered versus unlogged forested tracts (Martin and Grotefendt 2007).

Trees in RMZs with adjacent clear-cuts may be more susceptible to windthrow. Along non-fish-bearing streams, windthrow in RMZs (26m wide on both side of streams) increased instream large wood by 34% within one to three years after logging (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 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 activities 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).

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 with 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). Refer to Section 5.3.4.2 for additional information on impacts to riparian areas by livestock grazing in eastern Washington.

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 CONCLUSION

The 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. The main uncertainties for management are: 1) the shape of the wood recruitment function (e.g., Figure 5.4), and, in particular, 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.

With respect to in-stream wood, the management objective is to maintain in-stream wood sizes, abundance, and tree species that are within the historical range of natural variability. Given the current uncertainties, we believe that maintaining or re-establishing riparian forest conditions similar to unmanaged, historical conditions are the most reliable ways to provide in-stream large wood sizes and abundance that are similar to historical levels.

The most important management question is what riparian buffer width will achieve the management objectives for in-stream large wood? 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 In Washington State, Fox (2003) found that mean heights of canopy trees in old-growth riparian areas (i.e., stand > 200 years old) ranged from 100 to 240 ft Height variation among riparian areas was largely explained by forest zone and site productivity class. If large wood has a minimum diameter of 10 cm, then only that portion of a tree's stem greater than 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 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, depending on site productivity class. Therefore, to maintain 100% of a site's large wood recruitment potential, the riparian buffer should be about 85 to 230 ft wide, depending on site productivity class. This width only accounts for trees recruited via bank erosion, windthrow, or tree mortality. It does not account for 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.

The second most important management question is what should be the composition and structure of the riparian buffer? To meet the aforementioned management objective, the composition and structure must be similar to that of an unmanaged riparian forest.

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CHAPTER 4. STREAM TEMPERATURE

Kirk L. Krueger and Daniel J. Isaak

4.1 INTRODUCTION

The great diversity of Washington's landscapes creates equally diverse stream thermal conditions (Figure 4.1). Broad ranges in elevation (0–4,000 m, 0-13,000 ft), precipitation, stream size, topography, and other components contribute to some streams having average summer temperatures as low as 4°C (39°F); whereas others only a few kilometers away may exceed 20°C (68°F). 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, riparian vegetation and its condition play important roles 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; Caissie 2006; Boyd and Kasper 2003) 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 source of heating, reaching a stream (Poole and Berman 2001). Indirect effects of riparian vegetation include, but are not limited to, maintenance of complex channel form and 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, 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 work to describe the effects of human activity on stream temperature has focused on summer maximum water temperatures at specific sites adjacent to riparian and upland management. 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 reliably 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., Bowler et al. 2012; Czarnomski and Hale 2013; Elmore and Kauffman 1994; May 2003; Moore et al. 2005; Sather and May 2007).

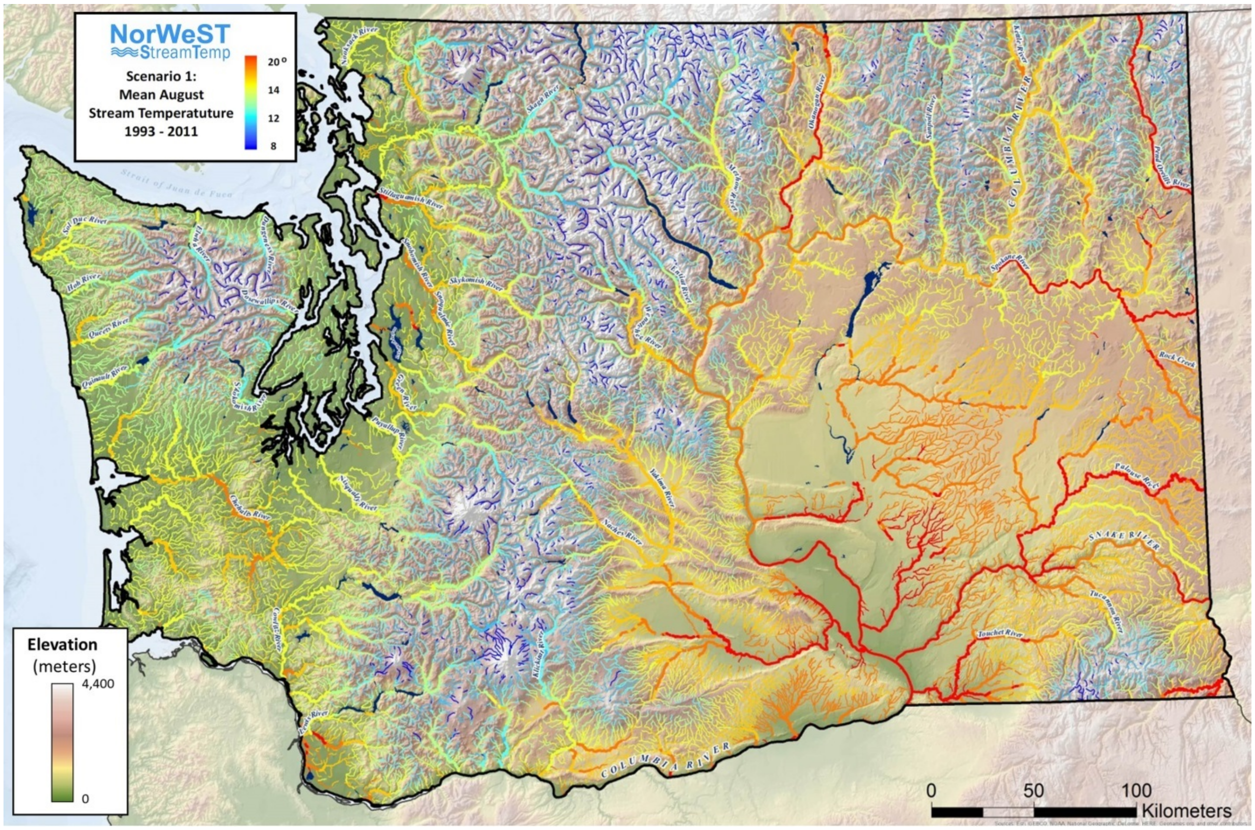

Figure 4.1. Stream thermalscape showing mean August temperatures for 66,236 kilometers (41,157 miles) of streams across Washington that was developed in the NorWeST project (poster available here: http://www.fs.fed.us/rm/boise/AWAE/projects/stream_temp/downloads/14NorWeST_WashingtonStreamTemperatureMap.pdf). 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., Allen et al. 2007; Sridhar et al. 2004). The literature, derived primarily from descriptive (e.g., case study) and occasionally statistically predictive (e.g., generalization across many case studies) studies demonstrate 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 (Janisch et al. 2012; Johnson 2004; Kibler et al. 2013; Moore et al. 2005).

Variability in stream temperatures among locations (Figure 4.1) and among some study results is due to both temporally stable geomorphic 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. 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.

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., Allen et al. 2007; Boyd and Kasper 2003). 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 the needed spatial and temporal extents, durations, and frequencies (Leinenbach et al. 2013). Further, they can provide useful information on systems and processes that are not readily measured in 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 of 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. Studies that have attempted this (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 data on stream networks (Ver Hoef et al. 2006; Isaak et al. 2014) have been applied to large stream temperature databases (NorWeST website: http://www.fs.fed.us/rm/boise /AWAE /projects /NorWeST.html) and these have enabled relatively accurate predictions ($R^2 = 0.90$; RMSE = 1.0°C or 1.8°F) 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, the models do provide accurate thermal status maps for context and could be used with mechanistic models or field measurements describing degree of alteration 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 (McCullough et al. 2009; Ward and Stanford 1982; McCullough 1999; Caissie 2006; Hinch et al. 2012). For example, the collapse of the Fraser River, B.C. 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 such differences may be associated with where local temperatures occur relative to the thermal niche of individuals 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; Wenger et al., in review). 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 components, structures 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 (i.e., potential heat energy that can be added) and stream thermal sensitivity (i.e., susceptibility to temperature change) 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 information is 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 (Cole and Newton 2013; Cristea and Janisch 2007) and watersheds (Bisson et al. 2013; Booth et al. 2014; Coffin et al. 2011), suggesting that our understanding of the composition, structure and processes that affect stream temperatures are well understood and can be 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; the amount of direct solar radiation reaching a stream via management for riparian shade and water flows via water storage, extraction, and return (Olden and Naiman 2010).

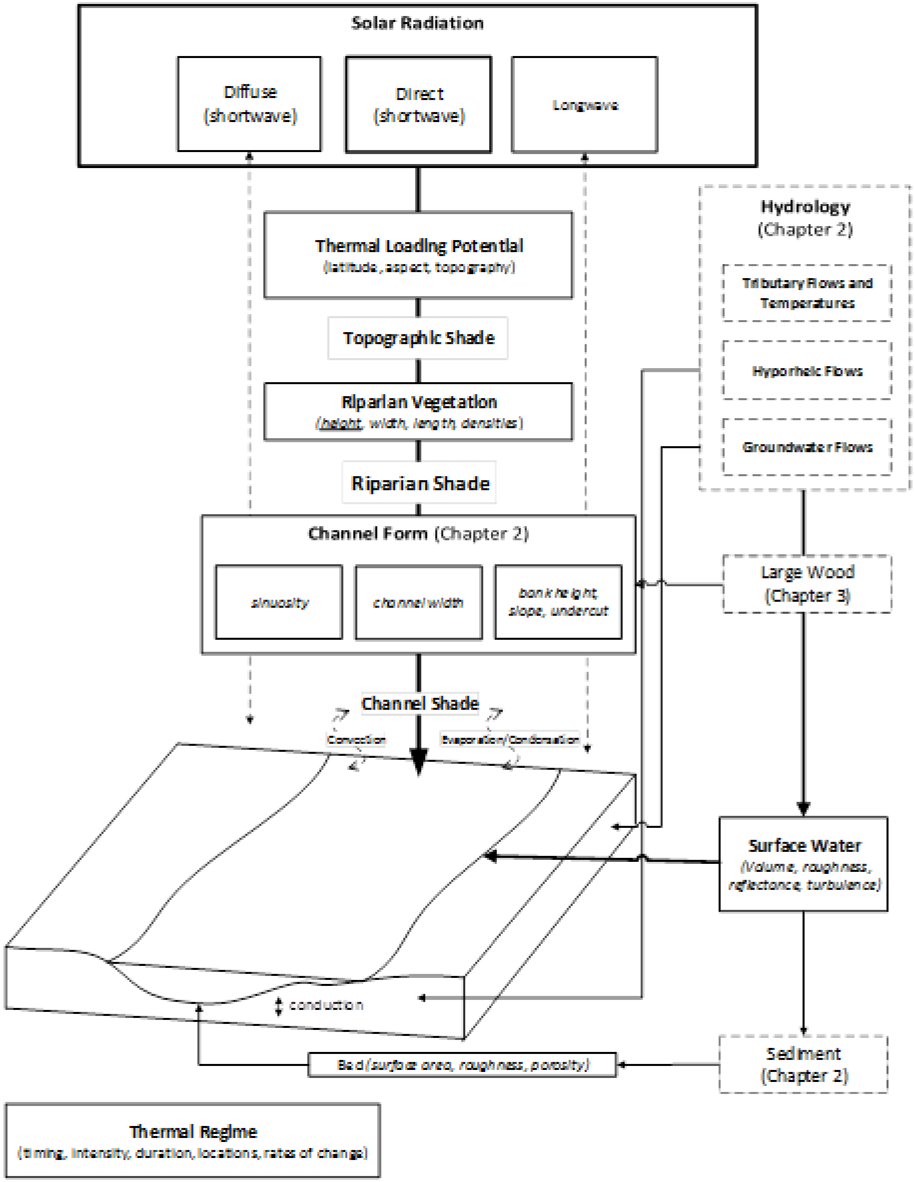

Figure 4.2. Conceptual model that identifies some factors that affect the thermal regime of streams. Relative importance is noted by line weight and arrows indicate directions of flows. Note that most of the elements in the model are influenced by human activities and that many components and structures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

The thermal loading potential of a stream reach is controlled in part by temporally stable and largely independent factors such as latitude, aspect, elevation, and topographic shading (Figure 4.2, Boyd and Kasper 2003). Streams at high and low latitudes are less exposed to solar radiation; the ultimate source of heat energy to streams, than streams at mid-latitudes. Stream aspect affects stream temperatures because the stream's orientation relative to the path of the sun (which differs seasonally) determines the amount of direct solar radiation that the steam 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. Latitude, elevation, aspect, and topography are temporally stable; they change on geologic time scales and result in differences in thermal loading potential among different streams and stream reaches (Boyd and Kasper 2003; Janisch et al. 2012). The effects of these 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., precipitation changes due to 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., Allen et al. 2007; Beschta 1987; Bowler et al. 2012; Caissie 2006; Garner et al. 2014; Johnson 2004; Poole and Berman 2001; Moore et al. 2005; Sridhar et al. 2004). 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 to volume ratio that can intercept and accumulate more direct solar radiation (Boyd and Kasper 2003) and also exchange more heat through atmospheric conduction. The width of the stream also affects its thermal sensitivity because the amount of shade that can be provided by topography and riparian vegetation is affected by stream width (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 (Bisson et al. 2006; Frissell et al. 1986). 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 2004) and affect channel form (Chapter 3; Abbe and Montgomery 1996; Keller and Swanson 1979; 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 (Story et al. 2003; Boyd and Kasper 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 important volumes of water at different temperatures to receiving stream water (Bobba 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 conduction (Sinokrot and Stefan 1993), by regulating the amount and rate 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 (Wondzell 2012; Wright et al. 2005). Hyporheic flows often have little effect on average stream surface water temperatures except for small streams at low flows and can create discrete locations of cool water (Wondzell 2012) that may serve as important as refugia for salmon. Hyporheic flows can be influenced by flow regulation (e.g. dams) that can reduce variation in both flows and temperatures, which reduces the potential for hyporheic exchange to act as a thermal buffer (Nyberg et al. 2008; Poole and Berman 2001; Ward and Stanford 1995).

The interaction of complex in- and near-stream riparian ecosystem processes can result in emergent properties (i.e., new rates that are not displayed by considering each process alone) and new system states (Allan 2004; Pringle and Triska 2000). These emergent properties 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; Gomi et al. 2004; Micheli and Kirchner 2002) 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 results 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 well understood, but complex due to the variability of the environment in which these processes occur (Figure 4.2). The relative importance of many of these components and structures 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 zone (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 riparian forest 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 who attempts to use the statistical model in Figure 4.3 to predict how a reduction from a fully forested 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°, and for another study between -0.5 to nearly 5 C°. 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 in temperate streams during all seasons except winter (Figure 4.4). 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 ecosystem 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 rates of metabolic processes and 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 (Eliason et al. 2011; Ayllon et al. 2013; Farrell et al. 2008). 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).

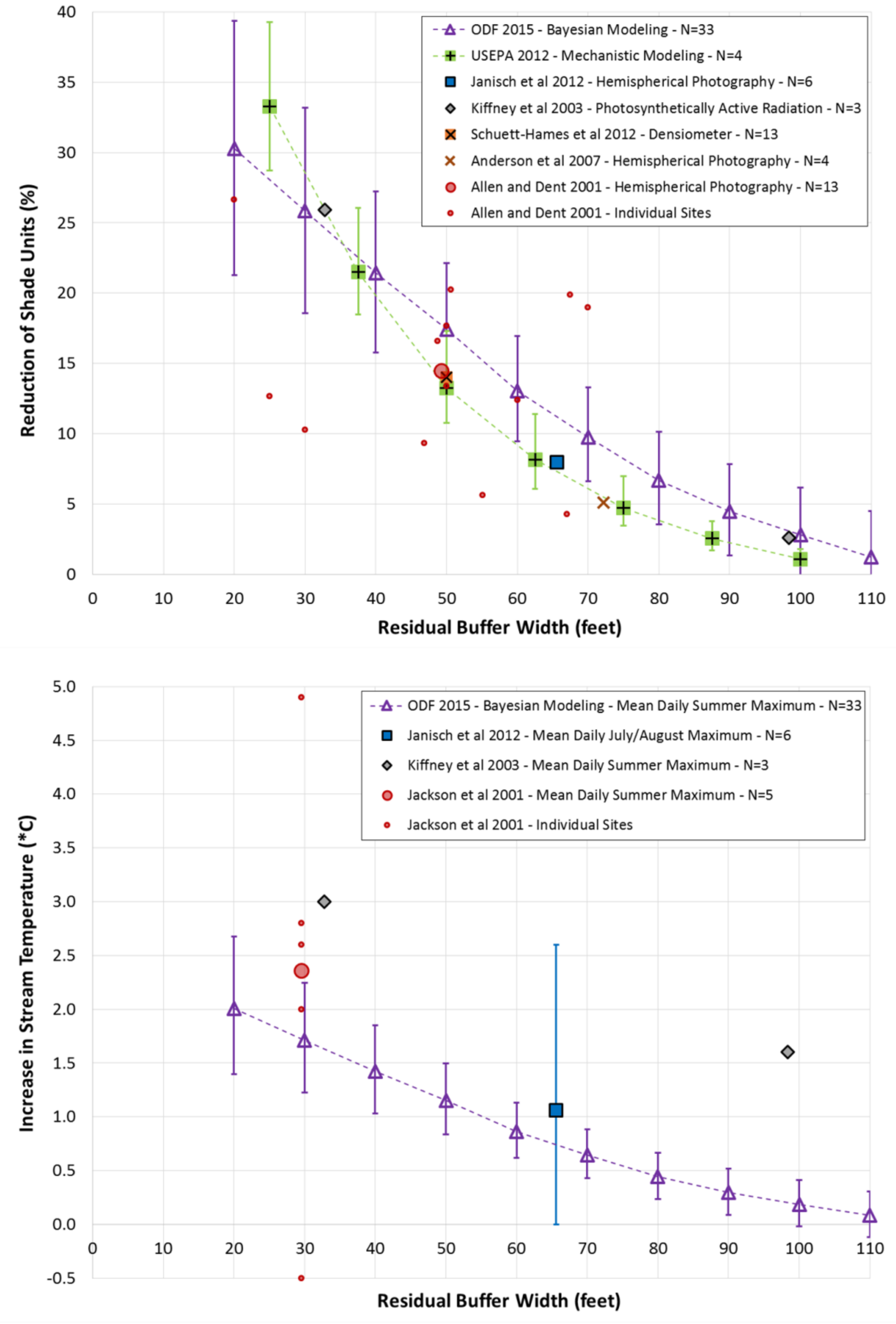

Figure 4.3. Observed shade (a: top panel) and temperature response (b: bottom panel) associated with "no-cut" riparian buffers with adjacent clearcut harvest. Only studies that employed a Before-After-Control-Impact design and conducted in Pacific Northwest forests are included. Shade Unit is percent 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.

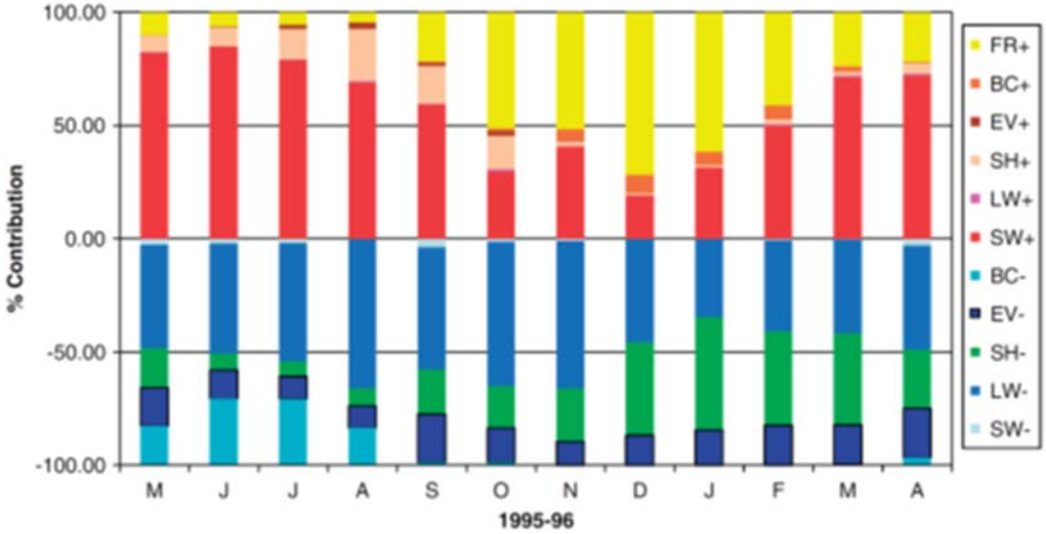

Figure 4.4. Annual cycle of mean monthly relative contributions of non-advective heat fluxes in the Black Ball Stream, Exmoor, South-West England, a temperate zone stream (adapted from Webb et al. 2008). The red portion of bars show the contributions of shortwave radiation to stream heating, which are especially large during spring and summer seasons when solar angles are largest and shading from riparian areas most important. SW = shortwave radiation, LW = longwave radiation, SH = sensible heat, EV = evaporation and condensation, BC = bed conduction, FR = friction.

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 (Eliason et al. 2011; Ayllon et al. 2013; Farrell et al. 2008). 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; Eliason et al. 2011; Farrell et al. 2008). 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. (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 the complex links between the physiology, behavior, environment, and 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 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.

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 of posed by altered thermal regimes on the survival and productivity of populations of salmonids or non-salmonid fishes, and on 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 when fry emerge and when their preferred forage is most 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. 2007). Altered thermal regimes can also affect ecological interactions. For example, Lawrence et al. (2014) demonstrate that higher stream temperatures can expand the geographic distribution of non-native smallmouth bass 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 Knutsen 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 that are important to management. In particular, some recent studies are able to measure thermal exposure through time, occasionally in field conditions, and estimate physiological attributes such as growth rate (Sauter 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 identifies 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 habitat (Fausch et al. 2002; Ward et al. 2002; Wiens 2002, Chapter 8), that is, they move between discrete locations of different temperature (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). Wise 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 riparian ecosystem management, altered thermal regimes, or thermal tolerance of 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 has 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 2008), while stream temperatures in which these species are found are considerably lower than those maxima (invariably ≤16ºC; e.g., Brattstrom 1963; Welsh 1990; Welsh and Lind 1996; Pollett et al. 2010). Moreover, water temperatures at which these species 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 lower 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 more elevated 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 20°C or 68°F, 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*, but 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 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 their 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 riparian ecosystem management, altered thermal regimes, or thermal tolerance of 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) and changes in emergence timing and asynchronous emergence among sexes due to small increases in summer temperatures have been reported (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 thermal regime has received little study especially 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 important 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 (Webb et al. 2008; Zoellick 2004). 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., riparian 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).

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 increases stream temperature due to forestry practices (Olson et al. 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 temperature 1.5 to 4°C (3 to 7°F) 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 and 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. While the importance of thermal regimes to conservation of fish and wildlife in rivers and streams is becoming clearer, empirically informing those connections through studies is challenging and thus rare. These challenges are more due to the scale and scope of the issues rather than the understanding of the mechanisms involved (Boyd and Kasper 2003, Poole and Berman 2001). For now, conservation efforts for maintaining stream thermal regimes must be based on our understanding the of mechanisms of stream heating affected by human activities, wise use of statistical models related to shade and stream temperature, and the need for connectivity of the watershed for fish and wildlife.

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 of the functions that affect stream thermal regimes among locations, 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, natural 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 importantly 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).

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 stream structure remains relatively stable (e.g., stream 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.

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 ground, hyporheic and surface water flows on steam 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 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 components, structures 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 (D'Souza et al. 2011, Gecy and Wilson 1990). 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 (*Oncorhynchus tshawytscha*) 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, component 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 size) 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 organism 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.

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CHAPTER 5. POLLUTANT REMOVAL

Jeffrey L. Ullman and George 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. 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 authoritative 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 POLLUTION REMOVAL FUNCTION OF RIPARIAN AREAS

Water quality refers to the physical, chemical, and biological characteristics of water, which serves as a measure of 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. 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 healthy 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 range 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.

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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. Absorption is the filling of minute pores in a substance by molecules or particles.

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. Following 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 and 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 through the riparian area, which influence natural chemical and biological processes that remove soluble pollutants such as sorption to soils, microbial degradation, and plant uptake (Vogt et al. 2012). Biogeochemical activity often increases as the groundwater flow approaches the stream margin,

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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.

which 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).

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 that can take up, assimilate, transform, and degrade contaminants. 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 (US DOI 2003). Soil particle size also tends to decrease moving away from the stream edge (Huggenberger et al. 1998). This spatial structural variability results in disparate hydrologic conditions, and corresponding differences in pollutant fate, at locations in close proximity.

The soil microenvironments described above influence hydrologic conditions within riparian areas, 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 similar 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 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 a more homogenous stand of an exotic species (e.g., Himalayan blackberry [*Rubus armeniacus*]) that often does not have the same vegetative density or root structure as the native population, thereby altering riparian area effectiveness in reducing pollutant movement to surface waters.

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 stream erosion. Soil stabilization also limits erosion and gully formation that can result from overland flow moving through a riparian area. In addition, forested riparian 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).

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 buffer systems 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.

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 (Au et al. 2004; Redding et al. 1987). 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 (WDOE 2007). 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 healthy cover of herbaceous litter and

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[5] Streambank erosion, channel incision, and surface erosion in riparian areas present other potential sediment sources. As this chapter is focusing 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).

woody debris. As mentioned in the vegetation section of this chapter, increased surface roughness increases friction, slows surface flow, and enhances infiltration promoting the settling out of sediments. 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.

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). Vegetative filter strips in the range of 3 to 12 m (10 to 40 ft) typically remove 50 to 90% of the sediment load (Leeds et al. 1994; Yuan et al. 2009), although greater widths have been recommended depending on the circumstance (Castelle et al. 1994). A significant body of research has shown that larger particle sizes are deposited within the first 3 to 10 m (10 to 33 ft) of a riparian buffer with smaller particle sizes settling out at greater distances (Sheridan et al. 1999; Syversen et al. 2001; Dosskey et al. 2002; Yuan et al. 2009). Correspondingly, the relationship between buffer width and sediment attenuation is non-linear, necessitating disproportionate increases in width to achieve an incremental increase in sediment removal. 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) suggest 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 10 m (33 ft) would sufficiently reduce sediment transport to the adjacent stream, assuming responsible timber harvest practices. A review of published research on forested riparian buffers conducted by Sweeney and Newbold (2014) concluded that sediment trapping efficiencies of about 65% and 85% are typical for 10 m and 30 m (30 and 100 ft) buffer widths, respectively.

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 what are thought to be 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 of 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 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), 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)

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[6] In general usage "best management practice" (BMP) means a practice, or combination of practices, that is 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).

purported 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 vegetative 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. The 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 1 m (3 ft) 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, it was 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 attached to sediments 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 5 m (16 ft) 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 that buffer widths upwards of 10 m (33 ft) offer greater removal efficacy. Research over the past four decades has consistently recorded 60% to 90% phosphorus attenuation in buffers in this general range (Dillaha et al. 1989; Vought et al. 1994; Lee et al. 2003; Darch et al. 2015). Forested riparian ecosystems assessments have yielded less consistent results, with studies showing phosphorus retention 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.

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 efficient 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, that could enhance phosphorus sorption (Zhang et al. 2014).

The relatively new research topic related to phosphorus attenuation focuses on the 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 in through riparian areas (Zhang 2014).

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 (US EPA 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 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 diversity of crops, including orchards 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, representing a notable input in industrial and mining regions (US EPA 1980). These sources introduce copper into the environment as a nonpoint source pollutant, which differentiates to a significant degree 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 photocyle, 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 be more readily retained in riparian areas. Conversely, metals can stay in solution as more mobile forms that can 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 the metals in the biomass through detoxification mechanisms (Sarma 2011). These plants 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 either be used to mobilize metals to promote uptake via phytoextraction or be implemented 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 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 (charcoal used as a soil amendment) 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 the grass tissues 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 necessarily connote new compounds, but also includes 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 effects 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 plant, animal, or fungal 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. 2015).

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 the compounds to react and transform into new products. However, some fundamental information on organic compounds will allow an understanding of the information required to determine the impacts riparian buffer size and structure have on attenuating 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 log octanol/water partition coefficient (Log K_{OW}) and organic carbon partition coefficient (K_{OC}) all serve as predictors of a substance's potential for mobility, behavior and potential for attenuation.

Natural organic matter plays 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 provides a predictive tool that indicates 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 represent a substantial component of natural organic matter. Humic substances have long been recognized 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 convoluted and beyond the scope of this chapter, but their potential to affect K_{OC} values by up to an order of magnitude for a given system is salient to note when predicting riparian attenuation potential (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 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 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 type of bioremediation that occurs in the soil. 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 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 routes of a pesticide; this is essential information to consider when determining appropriate riparian widths for attenuation. Essentially, 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., greater than 1,000 L/kg). While the literature lacks in 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; Reichenberger et al. 2007; Lacas et al. 2005). 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 USDA-NRCS recommends a standard width of 15 m (50 ft) with the aim of achieving 50% effectiveness, but there is not much evidence to support this guidance (USDA-NRCS 2000). Another general recommendation for diffuse runoff suggests riparian widths of 10 m and 20 m (36 ft and 66 ft) for hillslopes of under and over 100 m (330 ft) 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, as Ucar and Hall (2001) succinctly summarized information on the use of windbreaks as a mitigation strategy, there is no general consensus about what constitutes optimum riparian buffer widths to protect water quality. That review reports that research has yielded buffer width recommendations that range from 6 to 45 m (20 to 150 ft) for ground application and 25 to over 1,200 m (80 to 4,000 ft) 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 7.6 m (25 ft) 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 3 m (10 ft) no-spray buffer reduced herbicide drift deposition to adjacent surface waters by 95%, and no measurable drift reached the stream at a width of 6 m (20 ft) as long as the wind-speed was below 25 kph (15 mph) 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, a general trend has been observed that evergreen species display capture efficiencies 2 to 4 times greater than deciduous trees (Ucar and Hall 2001).

5.4.5 *Pathogens*

Pathogens impair more waterbodies in Washington than any other contaminant (US EPA 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 vernacularly common to refer to them as pathogens when discussing water quality (shortening the phrase “pathogens and pathogenic indicator organisms”), and while technically an inaccurate use of the term that convention will be used in this section. The context in which the term is used should allow the reader to differentiate between whether the reference is to pathogens specifically or to the broader inclusion of 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 often presumed 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, research that is more recent 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 (Chaubey et al. 1995; Coyne at al. 1995; Entry et al. 2000; Schellinger and Clausen 1992; Walker et al. 1990). 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 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 pollution removal by riparian areas is referred to as removal efficacy, removal efficiency, or percent removal. These measures are usually defined as:

$$E_R = \frac{m_{in} - m_{out}}{m_{in}} \times 100$$

where m_{in} and m_{out} are the mass or concentration of the pollutant entering and exiting a riparian area, 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 four meta-analyses, but the majority was 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. One-variable models from the four meta-analyses predicted a wide range of buffer widths necessary for 90% removal efficacy: 10, 12, 23, and 52 m (33, 39, 75, and 170 ft) (Figure 5.1, Table 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 critically 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 5.3).

Three separate meta-analyses looked at the results of research on nitrogen removal. 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%. These two models predict that buffer widths of 24 and 133 m (80 and 436 ft) are necessary for 90% removal efficacy (Figure 5.2, Table 5.2). Separate meta-analyses 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. found that riparian buffers were nearly twice as efficacious in removing nitrogen from subsurface waters as 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 least efficacious at removing nitrogen. Zhang et al. (2010) did find a relationship with vegetation type and that buffers composed of trees remove more nitrogen than buffers completely or partially composed of grasses. Zhang et al.'s model that included vegetation type explained 49% of the variance in nitrogen removal results.

Comparing regression models for sediment and nitrogen removal (Figures 5.1 and 5.2) 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 is removed at roughly15 to 20 m (50 to 65 ft), 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 for disparate pollutants 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 5.4), and with the addition of vegetation as an independent variable, the model explained 47% of the variance. The one variable model, i.e., width only, predicts 90% (actually 89%) phosphorus removal with a buffer 30 m (98 ft) 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 20 m (65 ft) wide.

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 10 m (33 ft) wide, the width of the 90% prediction interval is 64%. Therefore, while the model predicts that a 10 m buffer will provide an average removal efficacy of 70%, the actual removal efficacy obtained from another study on phosphorus removal could be anywhere between 36 and 100%. 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.

5.6 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 the 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

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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%.

[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.

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 often required a simplified description of some mechanisms involved with pollutant attenuation in riparian areas, this chapter, nevertheless, provides the basic information needed for understanding this important ecological function.

The central problem faced by resource managers is determining the adequate riparian buffer width, structure, and composition 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 best way to increase pollutant removal, but this is clearly 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.

Despite the many scientific uncertainties, management decisions must be made. 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. While buffer width is only one of many factors affecting the removal of pollutants by riparian areas, research shows that width is generally the most important factor. Research also indicates that the second most important factor is the vegetative composition and structure in riparian areas. Based on their meta-analysis of the research, Zhang et al. (2010) concluded that a 30-m (100 ft) wide, vegetated buffer under favorable slope conditions (≈ 10%) removes more than 85% of all the studied pollutants (sediment, pesticides, N, and P). In theory, buffer width and vegetation could be optimized for site conditions so that riparian areas protect water quality while minimizing economic costs on landowners.

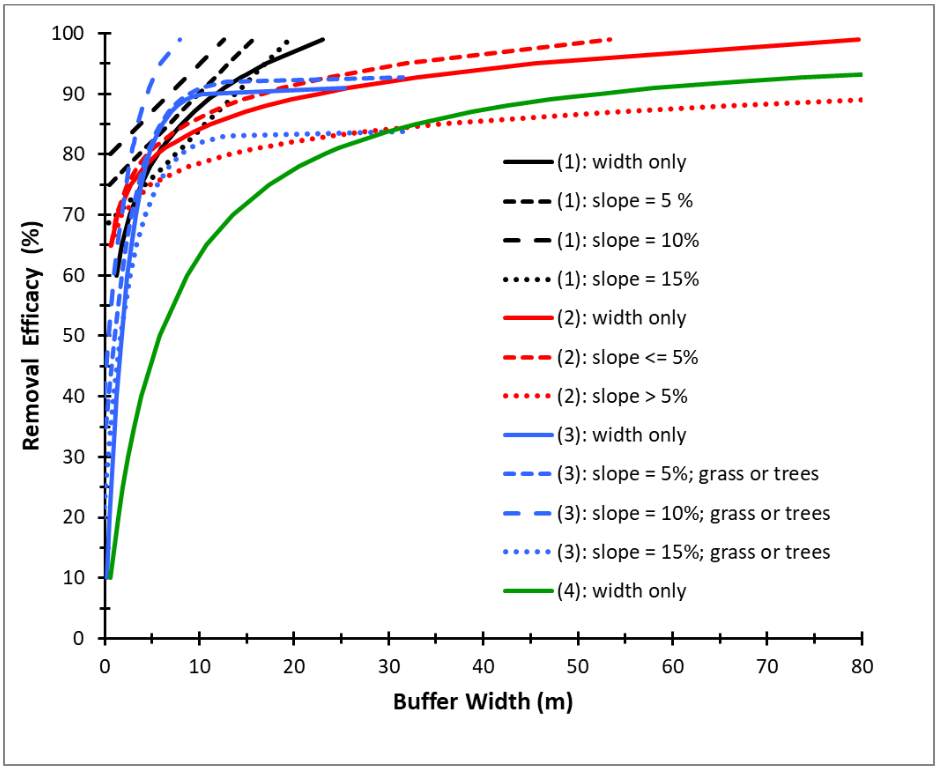


Figure 5.1. Function relationships for sediment removal in riparian buffers developed by four separate meta-analyses. Numbers in parentheses refer to different meta-analyses. See descriptive information in Table 5.1.

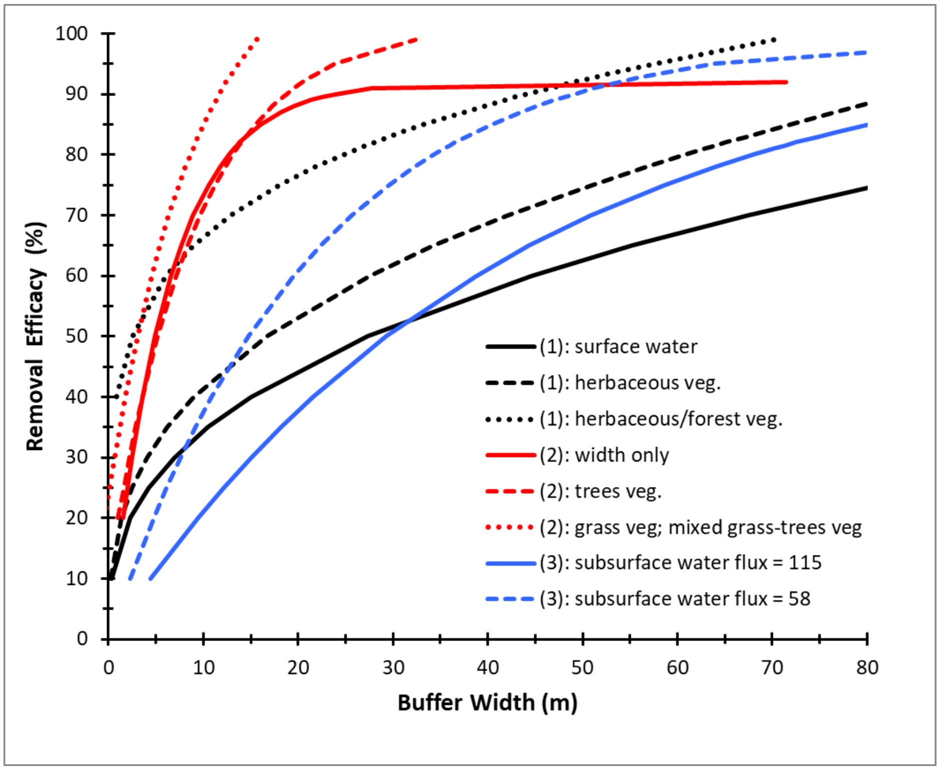


Figure 5.2. Function relationships for nitrogen removal in riparian buffers developed by three separate meta-analyses. Numbers in parentheses refer to different meta-analyses. See descriptive information in Table 5.1.

Table 5.1. Descriptive information for functional relationships on sediment removal shown in Figure 5.1 and nitrogen removal shown in Figure 5.2. Reference numbers 1 through 4 correspond to numbers in figures. In the equations, E_R is the removal efficacy, *width* is riparian buffer width, and *a*, *b*, *c*, *d*, k_{50}, and *q* (subsurface water flux) are parameters determined through linear or nonlinear regression.

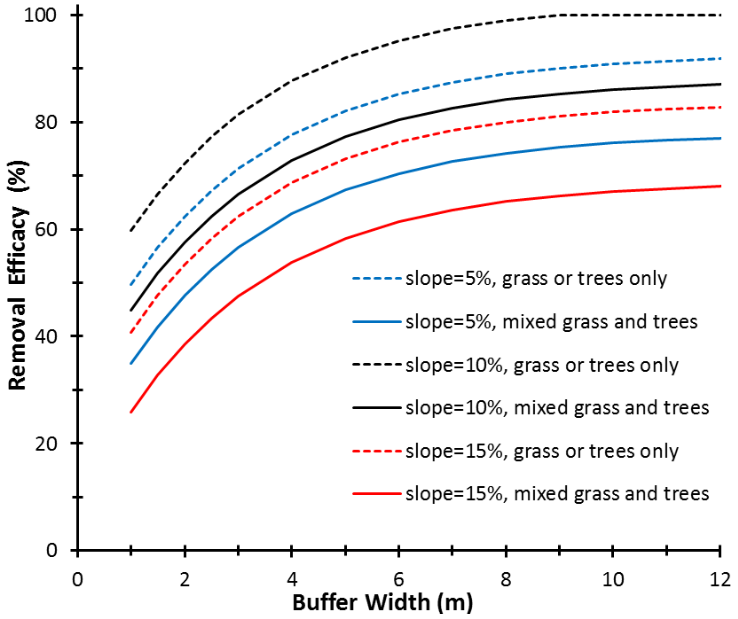
Pollutant	Meta-analysis	Form of Equation	Model	Nr studies	R^2	P value
Sediment	(1) Liu et al. (2008)**	$E_R = a + b\ \ln(width)$	width	79	0.34	<0.001
		$E_R = a + b \times width + c \times slope + d \times slope^2$	width & slope= 5%	79	0.43	< 0.0001
			width & slope = 10%			
			width & slope = 15%			
	(2) Yuan et al. (2009)	$E_R = a + b\ \ln(width)$	width	65	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
		$E_R = a\left(1 - e^{b \times width}\right) + c + d\ \times slope$	width & slope=5% & grass or trees	81	0.65	<0.001
			width & slope=10% & grass or trees			
			width & slope=15% & grass or trees			
	(4) Sweeney and Newbold (2014)	$E_R = \frac{width}{k_{50} + width}$	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 veg.	11	0.39	0.04
	(2) Zhang et al. (2010)	$E_R = a\ (1 - e^{b \times width})$	width	61	0.44	<0.0001
		$E_R = a\left(1 - e^{b \times width}\right) + c$	width & trees	61	0.49	<0.001
			width & (grass or mixed grass/trees)			
	(3) Sweeney and Newbold (2014)	$E_R = 100\ (1 - e^{b \times width/q})$	width & *q* set to 58‡	30†	0.37	na*
			width & *q* set to 115			

* signifies that P value was not reported in cited study.

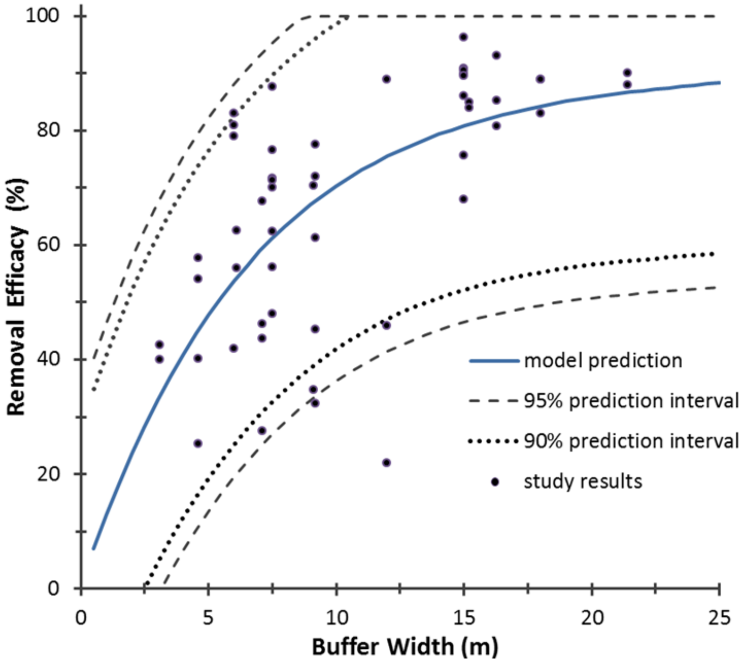
** Nearly all the studies included by Liu et al. (2008) and Zhang et al. (2010) in their meta-analyses were the same studies.

‡ 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).

† Studies for subsurface nitrate removal only.



5307 **Figure 5.3. A result from meta-analysis of Zhang et al. (2010). Sediment removal efficacy as a function of buffer**
5308 **width, site slope, and buffer vegetation (N=81, R^2=0.65, P<0.001).**



5309 **Figure 5.4. Meta-analysis of Zhang et al. (2010). Results of nonlinear regression for phosphorus removal efficacy**
5310 **as a function of buffer width (N=52, R^2=0.35, P<0.0001) with 90% and 95% prediction intervals.**

Table 5.2. Buffer widths in meters providing different levels of pollutant removal as predicted by meta-analyses. ">" symbol indicates that removal efficacy was outside the range of model but buffer width predicted to be at least as large as value in table. See Table 5.1 for description of sediment and nitrogen models and Figures 5.1 and 5.2 for graphical depiction of relationships.

			Removal Efficacy			
Pollutant	**Meta-analysis**	**Model**	**80**	**90**	**95**	**99**
Sediment	Liu et al. (2008)*	width	6	12	17	23
		width & slope =5%	4	10	13	16
		width & slope =10%	1	7	10	13
		width & slope =15%	7	14	17	20
	Yuan et al. (2009)	width	6	23	46	80
		width & slope ≤ 5%	5	17	32	53
		width & slope > 5%	13	98	267	596
	Zhang et al. (2010)*	width	5	10	>25	>25
		width & slope= 5% & grass or trees	4	9	>31	>31
		width & slope = 10% & grass or trees	3	4	6	8
		width & slope = 15% & grass or trees	8	>31	>31	>31
	Sweeney and Newbold (2014)	width	23	52	110	574
		median	5.5	12	28	28
		3rd quartile	8	22	40	69
Nitrogen	Mayer et al. (2007)	width, surface water only	97	133	153	171
		width & herbaceous vegetation	61	84	98	109
		width & herbaceous/forest veg.	25	44	57	70
	Zhang et al. (2010)	width	13	24	>30	>30
		width & trees	9	12	14	16
		width & (grass or mixed grass/trees)	13	19	24	32
	Sweeney and Newbold (2014)†	width & *q* set to 58‡	34	49	64	98
		width & *q* set to 115	68	97	127	195
		median	29.5	46.5	60.5	84
		3rd quartile	65	92	105	124
Phosphorus	Zhang et al. (2010)	width	14	>30	>30	>30
Pesticides	Zhang et al. (2010)	width	9	15	>30	>30

* Nearly all the studies included by Liu et al. (2008) and Zhang et al. (2010) were the same studies.

† 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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CHAPTER 6. NUTRIENT DYNAMICS IN RIPARIAN ECOSYSTEMS

Paul M. Mayer, Jana E. Compton and George 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, hydrologic flows, and animal movement. 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_2O 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 flow. 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 fate and transport 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. In the Pacific Coastal Ecoregion of Washington streams are generally oligotrophic, nutrient-poor, 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. Human activities to produce food and energy have increased the availability of N by at least three-fold and P by nearly four-fold (Falkowski et al. 2000; Sobota et al. 2013). While the human use of N and P benefits society, overuse also poses diverse threats to our health and well-being, ecosystem production, climate regulation, biodiversity, and aesthetics (Davidson et al. 2012). Excess N and P input to streams and estuaries may lead to harmful algal blooms, hypoxia, fish kills and contamination of drinking water supplies. Together, excess N and P are thought to cause billions of dollars of damage to freshwaters in the United States (Dodds et al. 2009; Sobota et al. 2015). Although stream nutrient concerns have a somewhat lower profile in the Pacific Northwest

(PNW) relative to other areas of the US, such as the watersheds of Chesapeake Bay or the Great Lakes, 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 (US EPA ATTAINS database). Numerous clusters of high nitrate in groundwater across Washington indicate excess N release into groundwater, where levels exceed 10 mg/L nitrate-N maximum contaminant level established for human health (Morgan 2016).

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 important 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 intra-soil 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 overland, subsurface 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). Soil compaction and impervious surfaces typically reduce retention capacity of nutrients in space and time by preventing infiltration and thereby limiting 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 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 nitrate N to gaseous N (Burt et al. 2002; Hefting et al. 2004).

A feature of streams key to nutrient dynamics is the area beneath the streambed and riparian area, termed the hyporheic zone, where surface water and ground water continually interact. This zone can have a large influence on chemical characteristics of the surface water and, therefore, play 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 nutrient uptake is inefficient because the system is saturated with nutrients and organisms can no longer use the incoming nutrient loads or because conditions are not optimal for uptake, such as when water flow is too fast for organisms to use, or because the stream does not support a community of organisms that effectively use the nutrients.

In 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 the 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, gas phase 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 symbiotic 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 (NO3-) and ammonium (NH4+), 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 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 and 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 animal wastes (Wise and Johnson 2011). Across most of the PNW, N exported by streams is derived from forestland, with N-fixation by red alder becoming important in forests of western Washington and Oregon (Figure 6.3). Farm fertilizer and livestock manure are most important for N in farmland and rangeland across the region. Point sources and urban runoff are the dominant sources of N near urban centers. For P, geologic sources dominate natural sources, with point sources, urban runoff, and farm manure and fertilizer important in areas under various 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 after timber harvest and burning. Timber harvest can increase sediment transport (Fredricksen et al. 1975; Binkley and Brown 1993), and forestry practices may remove vegetation in the riparian area and alter vegetation composition and condition. Roads in forested landscapes can be sources of sediment as well, although the type and condition of the road plays a moderating role on sediment delivery (Bilby et al. 1989; Luce and Black 1999). Urbanization also may contribute to degraded riparian ecosystem functions via increases in impervious surfaces leading to reduced subsurface flow, increased runoff, flashy stream hydrology, and increased N from leaky sewer pipes and septic systems (Paul and Meyer 2001).

Historically, a major source of allochthonous 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) estimated that the current biomass of salmon runs for all Pacific salmon species (Chinook, coho, sockeye, chum, and pink) in Puget Sound, the Washington Coast, and the Columbia River Basin are 25, 5, and 1 percent of historical biomass (circa early 1900s), respectively. This represents a loss of 122,470 US 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 Pacific salmon (Chinook, coho, chum) carcasses in streams decreased from 2,920 to 226 US tons, and that marine derived N and P decreased by more than 90%. Empirical evidence, presented below, suggests returning more salmon carcasses, and their marine-derived nutrients and energy, to riparian ecosystems could contribute significantly to the recovery of imperiled salmon populations.

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, the cycling loop and intra-system transfers missing without salmon are evident. 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., bears, otters, mink, bald eagles, ravens, and gulls) 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 watersheds of southeast Alaska derived approximately 23% of their foliar nitrogen from decayed salmon carcasses, and growth rates of Sitka Spruce 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, 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, beavers (*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). 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). Nutrient transfers can also be driven by deer or elk, or by livestock, which can be an important component of N and P delivery when livestock are allowed free access 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, governing temperature in part (Chapter 4) and largely control photosynthetic energy, 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 these nutrients are limiting in the PNW

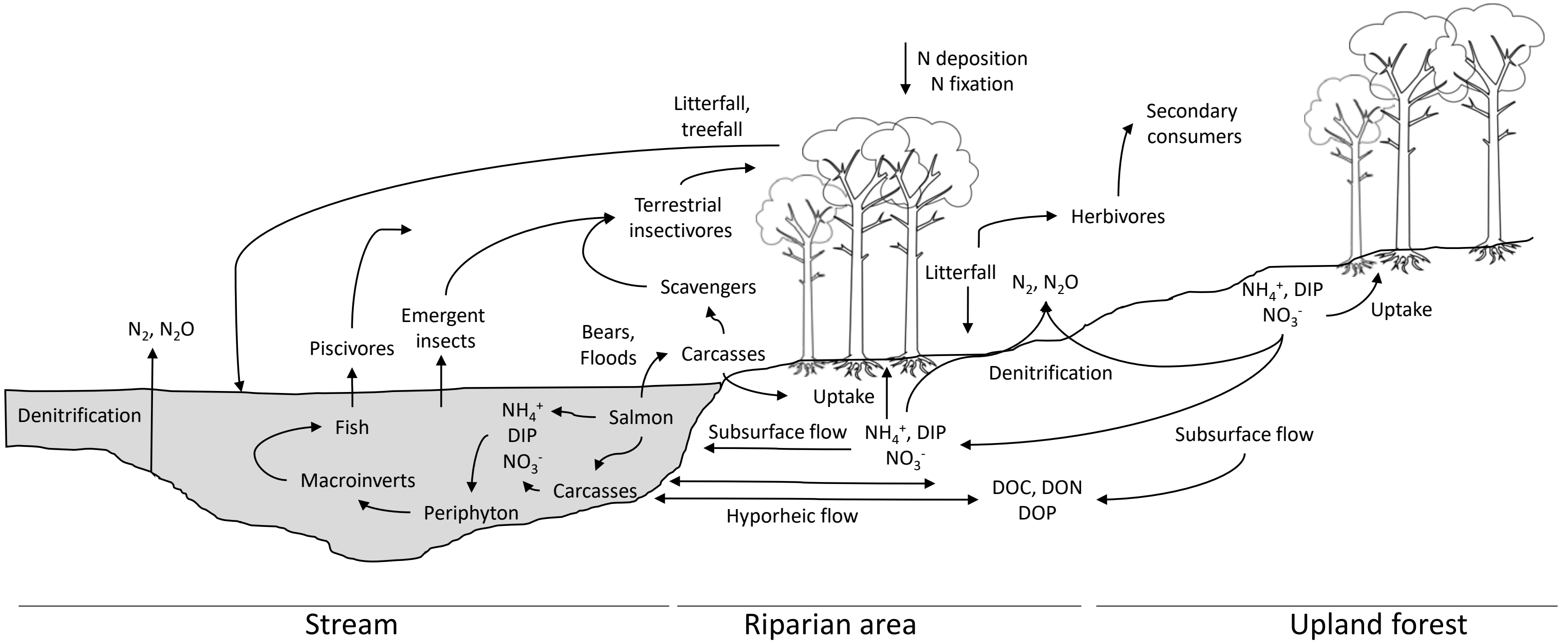

Figure 6.1. Nutrient cycle of an undisturbed PNW forested riparian ecosystem and connected stream and upland forest. Modified from Moyle et al. (2009). DIP = dissolved inorganic phosphorus; DOC, DON, DOP = dissolved organic carbon, nitrogen and phosphorus.

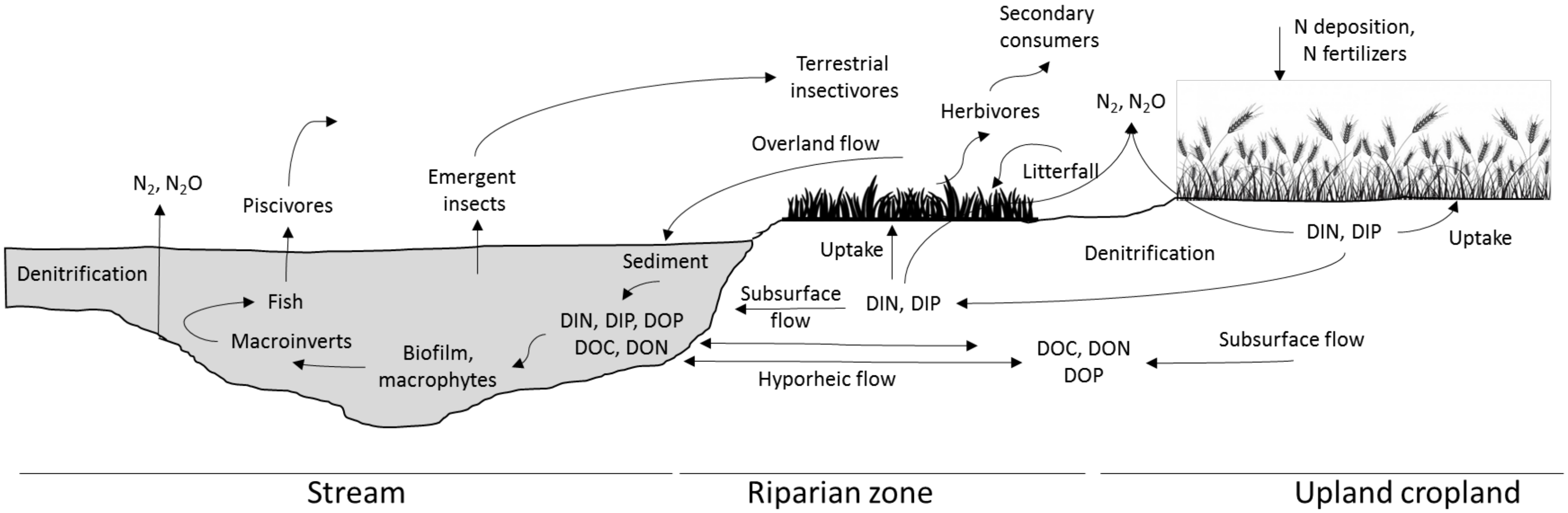

Figure 6.2. Nutrient cycle of a disturbed PNW riparian ecosystem and connected stream and upland cropland. Modified from Moyle et al. (2009). DIP = dissolved inorganic phosphorus; DIN = dissolved inorganic nitrogen (nitrate and ammonium); DOC, DON, DOP = dissolved organic carbon, nitrogen and phosphorus.

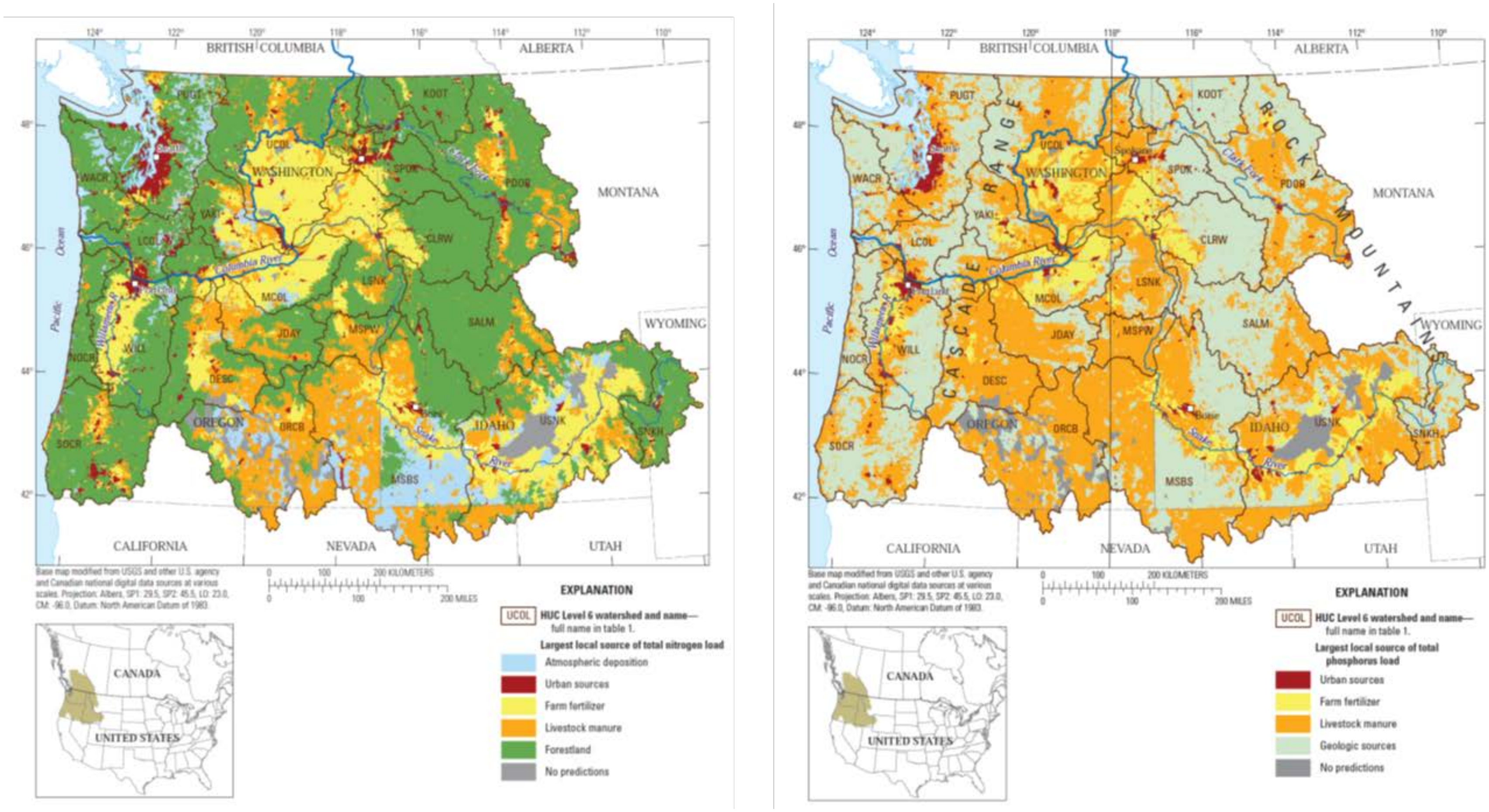

Figure 6.3. Dominant sources of total nitrogen (left) and total phosphorus (right) exported from incremental watersheds in the Pacific Northwest (based on 2002 data; Wise and Johnson 2013).

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 produced by different sources, thereby affecting its fate and transport. 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, 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 conditions (Mayer et al. 2010b).

Anoxic conditions also promote the production of methane. Production of methane and nitrous oxides, both powerful greenhouse gases increase in riparian areas with increased organic C supply (Kaushal et al. 2014). Dissolved organic carbon also can 1) increase the production of toxic disinfection by-products in drinking water supplies by reacting with chlorine to form trihalomethanes, 2) increase biological oxygen demand, thereby reducing oxygen for fish, and 3) adsorb metals and organic contaminants in streams and rivers (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 fate and transport 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 percent 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 by plant production, litterfall, decomposition, and hydrologic transport. 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). Because vegetation strongly influences nutrient quantity and quality in riparian systems, the structural roles of riparian vegetation are important considerations in riparian management plans.

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 function has 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, effects 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, p. 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 18.2 m to 25.3 m (60 to 83 ft) depending on a site's slope and wind exposure. The mean tree height of their three mature forest sites was 47.0 m (154 ft). Hence, riparian buffer widths needed to meet the 95% capture objective range from 39 to 54 percent 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 sp.*) females showed no upstream flight preference but those moving upstream contained twice as many eggs (Dudley-Williams and Williams 1993). In many cases, however, such as with baetid mayflies, 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 can affect nutrient cycling by slowing the downstream transport of water, sediment, organic matter and other materials (Chapter 3). Large wood that falls into streams traps sediments and modulates stream temperature through shading, influencing nutrient transport and cycling rates (Beechie and Sibley 1997). 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 also adds structure to the stream channel (Chapter 3) and creates obstacles that slow the flow of water extending the residence time of water in the stream 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 in substantial quantities 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[1] 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). The quality of leaf litter varies by species, with leaves containing more N, like deciduous red alder, being generally more nutritious than conifer needles. Litter may enter surface waters through direct fall, lateral movement along the ground, or mobilization during floods.

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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.

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 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. 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 alder litter can increase ecosystem productivity. For example, Wipfli and Musslewhite (2004) found that headwater streams with more 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 6.4). 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 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).

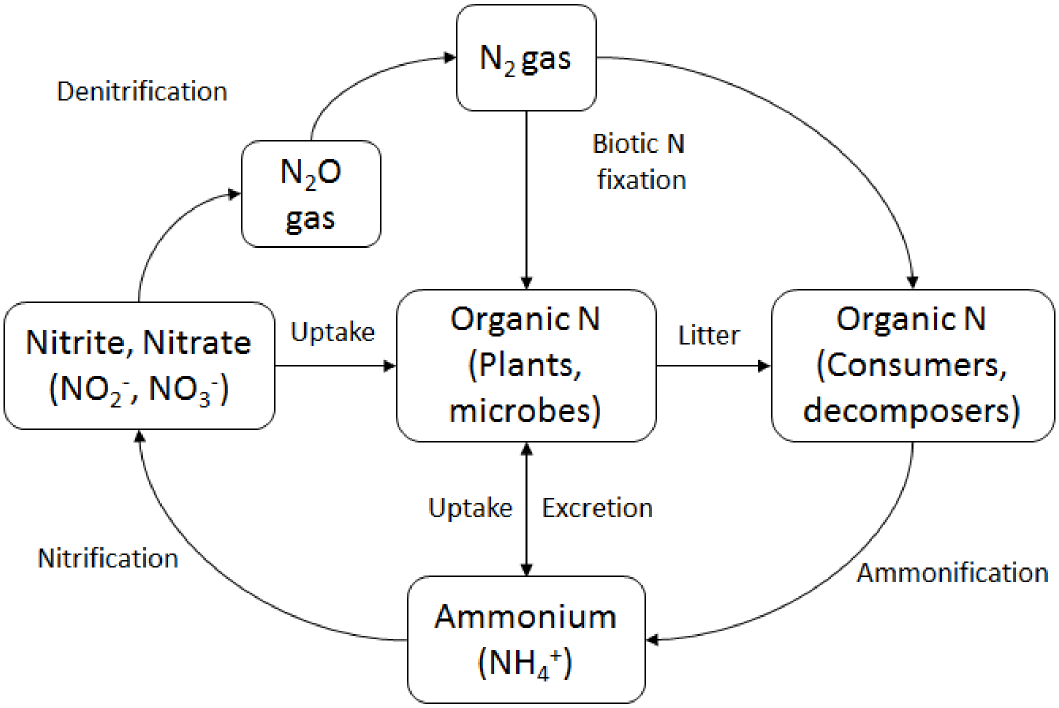

Figure 6.4. Nitrogen cycle pools and processes in ecosystems.

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 (>50 m; 165 ft) more consistently removed significantly higher proportions of N entering riparian areas than narrow bands of vegetation (0-25 m; 0-82 ft). 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 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 growing on the wood surface, which in turn is important in N uptake 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 is 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 N removal from regional groundwater prior to entry into surface flow (Hinkle et al. 2001), as well as important N removal and cooling as water moves in and out of hyporheic areas on its path downstream (Fernald et al. 2006).

The complex exchange of N among various storage components demonstrates the importance of maintaining diverse vegetation communities in riparian areas and protecting riparian soils and the natural pathways of water flow through the system. 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 component of 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 production 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 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 6.5). 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 (PO_4^{-3-}), and orthophosphate, are biologically reactive forms that 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 driver 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 relative to other sources under natural, undisturbed conditions has yet to be quantified for riparian ecosystems in the Pacific Northwest.

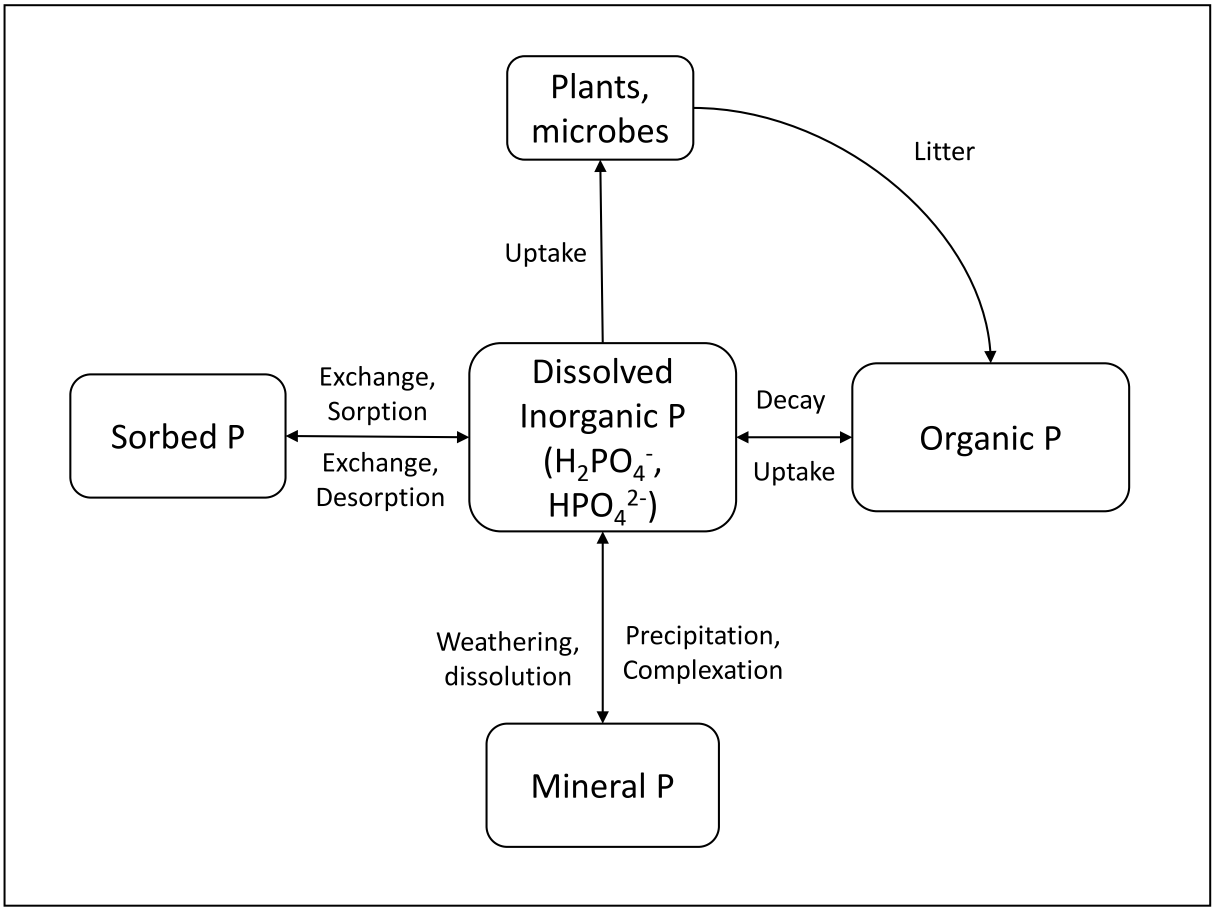

Figure 6.5. Phosphorus cycle pools and processes in ecosystems

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 orthophosphate to soil particles, may occur quickly (e.g. days) including adsorption and substitution between P and other anions on mineral surfaces, or 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 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 the 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 (Figure 6.6). 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 bring in nutrients from upstream areas. The hyporheic zone, where surface water and groundwater actively interchange, is the area beneath the streambed and extends laterally from the channel. As water moves downstream, it moves into and out of the subsurface, 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 large 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).

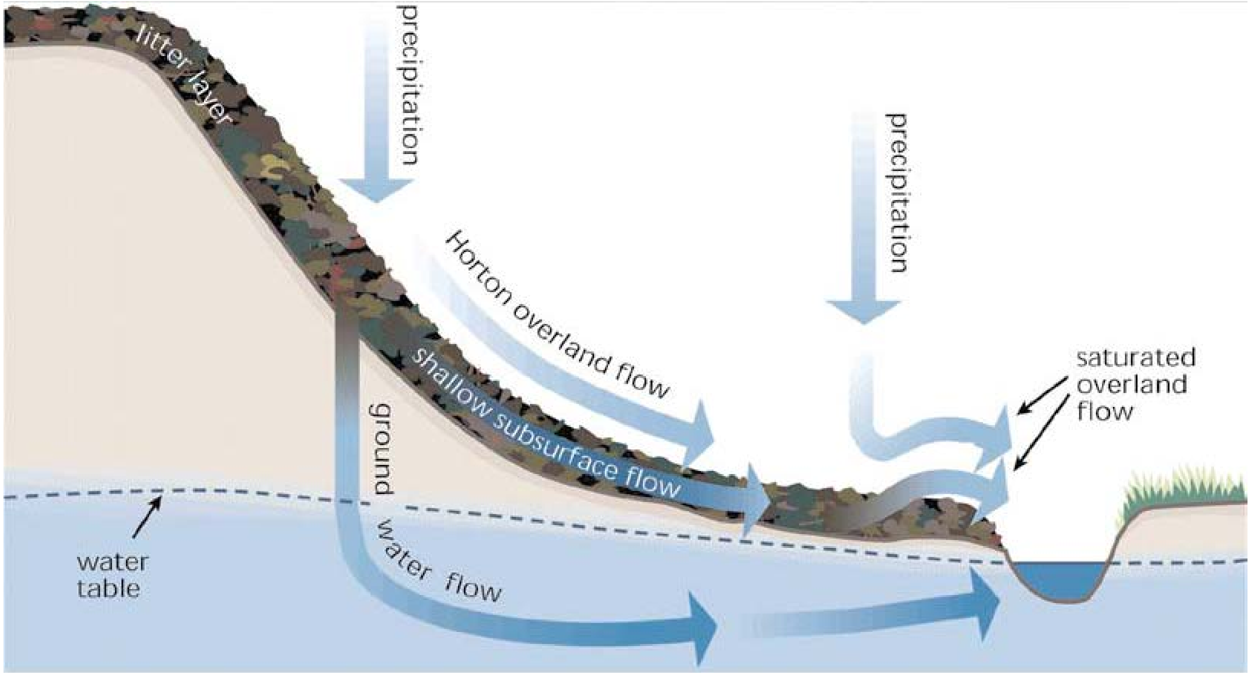

Figure 6.6. Scales of groundwater-surface water interaction (image adapted from Stream Corridor Restoration Handbook; FISWRG, 1998).

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 eroded and moved into the main channel. After floodplains are inundated, river water returns to the channel as overland flow, infiltrates the soil, or evaporates. Hydrology also impacts the rate at which N is released back into the atmosphere from riparian and stream ecosystems through microbial activity (denitrification) (Kaushal et al. 2014, Vidon et al. 2014).

Nutrient processing in streams is influenced by the size and characteristics of the riparian area and the hydrology of the system. 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).

Floodplains are active areas of nutrient exchange in riparian areas. In western Oregon and Washington, floodplains experience extremes in water and sediment flux (Naiman et al. 2010). Floodplains are dynamic, often having 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, natural flooding processes and nutrient exchange, both deposition and production, are severely altered often with negative consequences to riparian habitat including more severe flood damage if levees fail. 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 integrity and condition and to control erosion and nutrient runoff into streams (Mayer et al. 2007, Hoffman et al. 2009, Zhang et al. 2010, Sweeney and Newbold 2014).

Impacts to riparian systems 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 and are not well documented nor are they accounted for in current riparian assessments.

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 the 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) and because temperature has an effect on 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 reaches 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 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.

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 systems in the region are less well studied but there will be some common processes and dynamics. Understanding the flow dynamics and pathways is a key component of transferring information; in areas where subsurface flow paths bypass riparian areas, there is less potential for nutrient removal.

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., provision of 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 regulate light inputs in order to minimize algal blooms. However, stimulating algal and plant growth is one way to shorten nutrient spiraling lengths by enhancing nutrient uptake. Yet, this is a temporary feedback because nutrients will be returned after senescence. A longer-term goals of riparian management, is to avoid excessive nutrient delivery to streams. Maintaining riparian vegetation, soil conditions and flowpaths, etc. will enable the riparian area to remove nutrients from surface and subsurface flow before entering the channel. If nutrient input is successfully controlled, the issue of light levels promoting an algal bloom becomes much less of an issue.

Some of the critical gaps in our understanding occur where physical sampling for detecting water and material movement is most difficult, such as in the soil subsurface and in 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 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. 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 fate and transport 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 carcasses 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 the structures, such as large wood and processes that retain nutrients is 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.

Factor influencing riparian area	**Impacts of human activities**	**Role of management in reducing impacts**
Land use - agricultural	Excess nutrient application and leaching into groundwater and surface waters	Improved nutrient management can minimize nutrient loading - right time, right place, right source, right rate (4Rs)
	Compaction and vegetation damage from poor livestock management near streams	Riparian conservation or restoration may support attenuation of sediment and nutrient loads
	Riparian disturbance or lack of natural vegetation in riparian area	Riparian conservation or restoration may reduce light inputs that can drive algal overproduction and eutrophication
Land use - urbanization	Impervious surfaces increase nutrient and sediment transfer to suface waters	Restoration and management may focus on nutrient control
	Increased nutrient application through yard fertilizers and pet waste	Education and outreach to reduce nutrient loads
	Leaky and ineffective sewer and septic systems	Improvements to sewer systems, sewage treatment, storm drainage and septic systems
Land use - Timber harvest and logging roads	Small increases in nutrient loading	Alternatives to clear cutting, protection of riparian areas
	Small to moderate increases in sediment transport	Improvements to harvest practices, slope and soil considerations, better road construction, alternatives to clear cutting, protection of riparian areas
Stand age and composition	Recent clearcuts may lose the most nutrients	Mature stands likely retain the most nutrients
	Red alder fixes a large amount of N, this N remains in the system to support the productive soils of the Coast Range and western Cascades.	If stream nutrient levels are a concern, alder cover may be a factor.
Climate and seasonality	Rapid movement of water during fall and winter rains, sometimes bypassing riparian zones	Slow water movement across the landscape
	Rapid nitrogen cycling during the summer, but low flows means this inorganic nitrogen accumulates in the soil, fall rains displace this N and move it into streams in a pulse	Carefully consider fall fertilizer application
	Spring season usually has the peak algal blooms because nutrient supplies are high as is light availability before full leaf out.	
	Slower processing of nutrients in colder weather (although conifer trees can take up nutrients year round)	
Elevation and topography	Steep slopes lead to higher runoff and sediment transport	Forest and pasture management consideration of minimizing disturbance to steeper slopes
Hydrology	Reduced floodplain connection limits nutrient uptake	Long residence times can increase nutrient uptake and enhance ground water-surface water interaction
Nutrient concentrations, forms and inputs	High concentration, multiple sources, bioavailable nutrient forms may overload capacity of riparian system to process nutrient loads	Terrestrial vs atmospheric sources, careful management of synthetic fertilizer and manure application, care to balance N and P
Soil properties and geology	Low permeability reduces infiltration and interaction with soils	Rich, fertile soils can contribute organic carbon for denitrification
	Sandy soils can transmit more water and nutrients when applied (e.g. pumice soils and septic systems)	Some soils are better at adsorbing P (e.g., volcanic and highly weathered soils)
Other biota, macro and micro	Reduction in salmon runs reduces supply of organic matter for invertebrates, birds, mammals other fish	Diverse biota may increase nutrient retention: nutrients may be assimilated by vegetation, translocated by animals, or converted by microbial processes

6.5 SUMMARY

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 productivity, composition, and structure 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. However, N-fixation by red alder is an important component of nutrient cycling in forests of western Washington and Oregon. Wider riparian areas were more effective in N removal across studies. 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, terrain, 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.

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CHAPTER 7. RIPARIAN AREAS OF THE COLUMBIA PLATEAU

George Wilhere

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 1.1). In the driest parts of the ecoregion, a rain shadow cast by the Cascades Mountains limits precipitation to 6 to 9 inches per year, and even wettest parts of the ecoregion, the Palouse Hills Subregion, receive only 18 to 23 inches 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 we know about riparian areas in the Pacific Northwest is based on research conducted in forested ecoregions. During our review of the scientific literature, we found far less information about the structure, processes, and functions of riparian ecosystems in unforested ecoregions, such as the Columbia Plateau, than we found for forested ecoregions. However, many of the basic principles and qualitative relationships that have emerged from research in forested ecoregions should also be valid in unforested ecoregions. For instance, while we know there will be quantitative differences among ecoregions in the amount of wood or shade provided by riparian areas, we expect the physical processes of large wood recruitment and stream shading to be very nearly the same. On the other hand, processes that are significantly affected by groundwater, such as nutrient dynamics and pollutant removal, we expect to be different. To clarify our current understanding, throughout this chapter we 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, we make few subregional distinctions regarding riparian areas. Most of what we know 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

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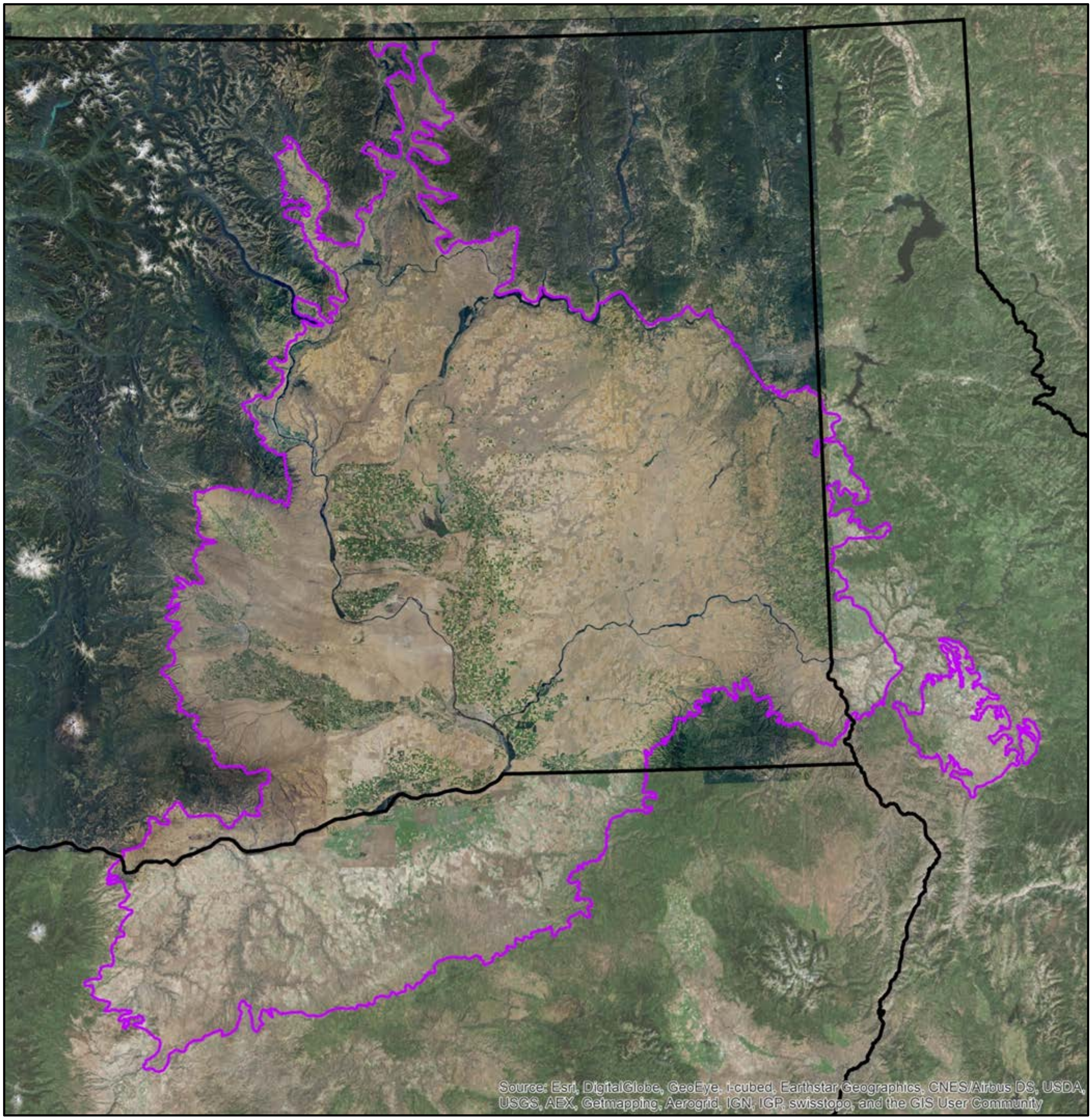
[1] There are several different delineations of ecoregions (e.g., USFWS 1994, Bailey 1995, Olson et al. 2001). For this report we use the ecoregions of Omernick (1987). The only non-forested ecoregion in Washington is the Columbia Plateau. Forested ecoregions surrounding the Columbia Plateau in Washington are the Eastern Cascades, North Cascades, 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.

Columbia Plateau, we believe many of the basic principles and qualitative relationships that have emerged from research in other desert ecoregions should also be valid for the Columbia Plateau.



Source: Esri, DigitalGlobe, GeoEye, i-cubed, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community

Figure 7.1. Boundary (in purple) of the Columbia Plateau Ecoregion. State boundaries are black. Aerial photography for Washington State done in 2015 by the National Agriculture Imagery Program.

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 1.2). In forested ecoregions, the potential vegetation of both zones is commonly forest, and the ecological processes and functions of the zones are similar. In the Columbia Plateau, vegetation within riparian ecosystems often exhibits an abrupt demarcation between the two zones. Trees, shrubs, and hydrophytic herbaceous plants are confined to moist streamside areas, but the upland zone of influence may consist of sagebrush or bunchgrass communities. Consequently, the processes and 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. For our purposes, the phrases riparian ecosystem and riparian area are synonymous.

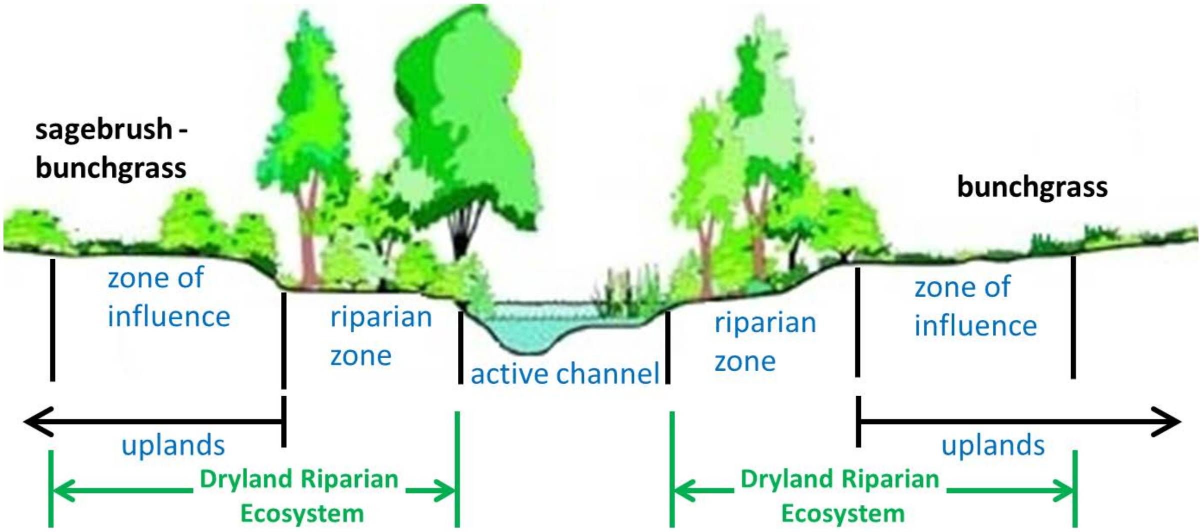


Figure 7.2. Dryland Riparian Ecosystem. The riparian ecosystem consists of two zones: riparian and zone of influence. The riparian zone extends from the edge of the active channel or channel migration zone towards the uplands. This zone includes areas where biota and microclimate are influenced, at least periodically, by flowing waters. Beyond this is the riparian "zone of influence." This includes areas where ecological processes significantly influence the stream, at least periodically. Note that many dryland riparian areas are treeless (diagram modified from USFS 2004).

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[4] streams generally

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[4] 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

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 Lierman et al. 2012). Furthermore, within the Columbia Basin the flow source of most streams is groundwater (Reidy Lierman et al. 2012), and consequently, during the hot, dry summer, many streams originating within the ecoregion are intermittent (Daubenmire 1942, Bryce and Omernik 1997). Spatial and temporal variability in local hydrology affects the distribution, abundance, and species of plants in dryland riparian areas.

The diversity[5] of riparian climax plant communities in the Columbia Plateau Ecoregion is greater than those of the 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 1.3; 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*).[6] 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 (yellow [*Salix lutea*], sandbar [*S. exigua*]), which can be up to 20 ft tall.

The structural diversity of riparian vegetation in the Columbia Plateau causes differences in site-level processes and 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.

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periods when it receives water from some surface or subsurface source, such as melting snow or a spring (Meinzer1923).

[5] Measures of biological diversity take into account two factors: 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.

[6] Black hawthorn might also be considered a tall shrub. Crawford (2003) reported black hawthorn in riparian areas to be 6 to 20 ft tall, and Daubenmire (1970) found black hawthorn up to 27 ft tall.

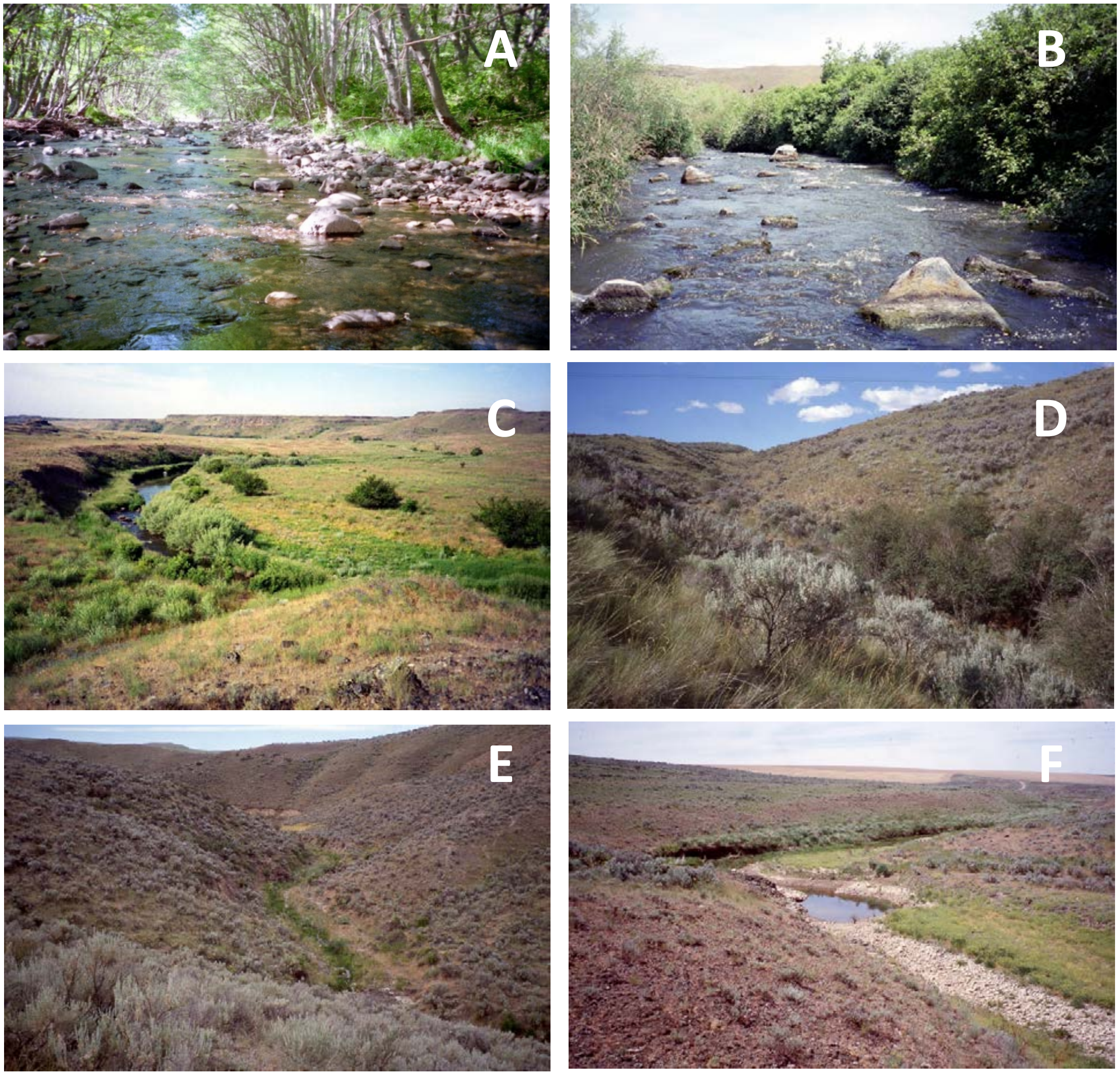

Figure 7.3. Different riparian vegetation structure types: A) tall tree, B) short tree, C) tall shrub, D) shrub on an intermittent stream, E) grass-like, F) forb. (Photo credit: Rex Crawford).

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 relatively 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 trapping, timber harvest), water management (dams, diversions, reservoirs), or invasive species (reed canarygrass [*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 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[7] 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 1150 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, we believe that 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 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

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[7] 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, WA. 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).

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 we discuss in Section 7.3.

7.2.2 *Open Range Grazing by Livestock*

Open-range grazing by livestock was the next major 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 Wally Valley in 1834 (Carlson 1940). By 1855, roughly 200,000 cattle inhabited the Columbia Plateau of Washington (Daubenmire 1970). Sheep raising 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" (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 indicate the density of woody plants that historically existed 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) believe this relationship can persist indefinitely within a given area. They also speculate 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, believes 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 during the absence of wolves 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 (henceforth, collectively known as wild horses). Until the mid-1900s, wild horses inhabited open rangelands throughout 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 nd). Wild horse populations also exceed 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. We have little historical 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) speculates that black cottonwood, the largest tree species in the Columbia Plateau, was much more abundant before the arrival of settlers.

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). We 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 are 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) reports 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.

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 of farmland (USBR 2017a, 2017b). Another 37,000 acres are irrigated with water impounded by dams on the Snake River (USACE 2002). All 1.17 million acres of irrigated farmland were once shrub-steppe or steppe vegetation. We do not know, however, 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 long. The reservoir behind Grand Coulee Dam, Lake Roosevelt, inundated roughly 84 miles of riparian habitat within the Columbia Plateau Ecoregion along the Columbia, Spokane, and Sanpoil rivers.[8] Roughly 460 miles of the Columbia River lie within the Columbia Plateau Ecoregion, but only 51 miles 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 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

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[8] Estimated with Google Earth. Distance estimated along Spokane River comports with that reported by Ortolano and Cushing (2000).

(Ortolano and Cushing 2000). In contrast, due to groundwater withdrawals, the groundwater table around upper Crab Creek (the Odessa subarea) has fallen 150 ft (KWA 2004). How this has changed stream flow in upper Crab Creek and its tributaries has not been studied.

Decades of hydrological alteration by dams and water diversions have altered vegetation composition and structure of riparian areas. Black cottonwood and willow (*Salix* spp.) 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 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 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) 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, we can confidently infer that historical conditions have been altered by regulated flows. If flow regimes are not changed, then as older individuals senesce and die, cottonwoods may be eliminated from riparian areas along 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 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, Easton, Bumping, Clear, and Rimrock lakes, large wood that could potentially interfere with dam operations is removed and burned (W. Meyer, WDFW, pers. comm.; B. Renfrow, WDFW, pers. comm.). 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.

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.[9]

Reed canarygrass 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 canarygrass 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 canarygrass 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 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*

Our current 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 multiple 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, cottonwood, and aspen. Cattle were said to be lost for weeks in these tangles. These same streams

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[9] 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*).

are now reduced to a single, incised thread with minimal riparian vegetation (P. Harvester, WDFW, pers. comm.). 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—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, we currently have a poor understanding of ecosystem structure, processes, and functions of riparian ecosystems. Historical reconstruction using journals of early explorers and naturalists, General Land Office survey notes, and historical drawings or photos (Figure 1.4) are ways to develop qualitative descriptions of riparian plant 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 limited value in establishing desired future conditions for restoration.

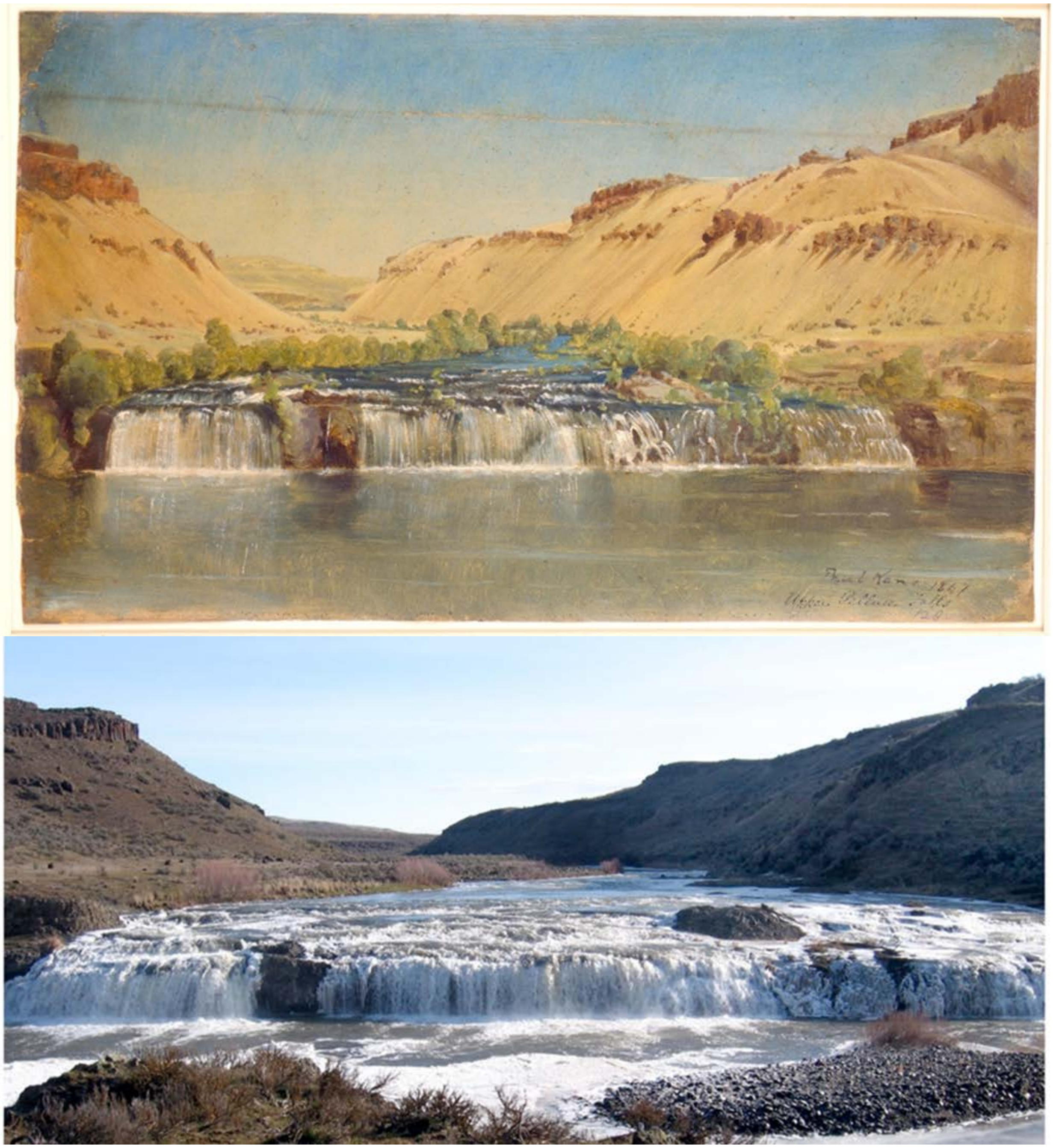

Figure 7.4. Historical drawings or paintings can provide invaluable information for restoration of riparian areas. A painting by Paul Kane done in 1847 of Upper Palouse Falls (top) shows extensive, uninterrupted small tree or tall shrub vegetation along the river banks. A recent photo (bottom) although taken in winter, shows large gaps in small tree or tall shrub vegetation along the banks (photo credit: Jack Nisbet)

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 long, 10 ft high, and stored 5 acre-feet of water. Scheffer (1938) implies that the dams reduced downstream flood damage and increased summer stream flows for irrigation. Finley (1937) reports the opposite situation for Silver Creek in southeastern 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 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 processes and 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 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 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 a significant predictor of 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 suggest 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 compete with or prey upon native fish.

7.4 ECOSYSTEM FUNCTIONS AND PROCESSES

FEMAT (1993) identified five key functions (or processes) 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 we know 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 our understanding about riparian areas in forested ecoregion is transferable to riparian areas in the Columbia Plateau, and therefore, where appropriate we 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.

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, we 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 Lemmons'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 forest, and a forested community that exists within 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 forested 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 5 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 2 times greater root biomass than incised channels. Along unincised channels, approximately 40% of root biomass was deeper than 20 cm in the soil, but along incised channels less than 25% of root biomass was below 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.

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 soils are unsaturated. Matric[10] 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).

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.

We could find no studies conducted in the Columbia Plateau describing the density of trees or their spatial distribution in riparian areas. However, we are certain that riparian areas in the Columbia Plateau contain less large wood than forested ecoregions, even historically, for three reasons. First, trees are smaller in the Columbia Plateau. Canopy dominant trees in old-growth riparian ecosystems of humid forested ecoregions can be well over 48 inches dbh and over 150 ft tall, but the maximum size of black cottonwood in the Columbia Plateau is much smaller (see below).

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[10] 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).

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 but 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 play a much smaller role in the fluvial processes of dryland river and streams.

Our knowledge of 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, 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 inches in diameter and greater than 6 ft in length.[11] 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 diameter at breast height (dbh) of about 30 to 35 inches 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 inches dbh and 52 ft tall at age 80 years, and 26 inches dbh and 70 ft tall at age 150 years (Roe 1958). Other tree species that might also 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 inches 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 3-8 cm (1.2-3.2 in.) in diameter (Collen and Gibson 2001), and a large proportion of woody stems used as food are also used in dam

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[11] 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 inches in diameter and greater than 3 ft in length. Small wood is also known as small woody debris or fine woody debris.

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 10 cm (4 in.) 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.

We expect that large wood recruitment processes in riparian areas of the Columbia Plateau will be very similar to those in forested ecoregions. 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. Although we have found no studies on recruitment mechanisms of small wood, we expect the main mechanisms to 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 transferred to an aquatic system is also called an allochthonous input of nutrients.[12] 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. We discuss detrital nutrients in the larger context of nutrient dynamics (e.g., nitrogen cycle or mass balance) when relevant information is available.

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). We 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 litter. Artificial fertilizers are used

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[12] Allochthonous means originating or formed in a place other than where found. In contrast, autochthonous means originating or formed in the place where found.

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 obtain in dryland riparian areas. Hence, in addition to providing nutrients, litter from riparian vegetation may also increase 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 area 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). Consequently, the main source of energy is allochthonous chemical energy from riparian areas. In arid and semi-arid regions, the canopy over headwater streams is 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*)[13], 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 350 $g/m^2/yr$ at Rattlesnake Springs and 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

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[13] Watercress (*Rorippa nasturtium-aquaticum*) is an exotic plant in Washington State.

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 we 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 (*Salix* spp.) 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, we focus on shade provided by vegetation. Vegetation also affects local air temperature, 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 can lower water temperatures of individual pools (Ebersole et al. 2003b, Ebersole et al. 2003a).

A rudimentary classification system based on overstory conveys the obvious 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. We do know 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. We lack a mapping of current and potential riparian vegetation types across the Columbia Plateau that would enable us to assess impacts on fish-bearing streams from insolation (but 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"[14] are sediments, excess nutrients (nitrogen or phosphorus), pesticides, and pathogens.

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, or 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, we focus on vegetation.

Research on pollutant removal by riparian buffers conducted in arid or semi-arid ecoregions is very limited (Hook 2003). We 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 buffers removed at least 94% of sediment regardless of vegetation type in the buffer, but we do not know whether the simulations are realistic 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 wide buffers removed 75 to 80% of sediment during a simulated 50-year 24-hour storm event. However, while the buffer was 40 ft wide, the bare cropland was only 15 ft 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 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

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[14] 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.

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.[15] We found 5 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). See Chapter 5 for our complete summary of their results.

The number of studies in the 5 meta-analyses ranged from to 52 to 88. We could determine the geographic locations of all studies included in 4 of the meta-analyses; only 1 study was conducted in an arid or semi-arid region (i.e., Tate et al. 2000 in Zhang et al. 2010). Four meta-analyses analyzed sediment studies, 3 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). Taking the median value of 4 meta-analyses (width only models), a buffer 57 ft wide is needed for 90% removal of sediment and buffer about 2 times wider is needed for 95% removal. Taking the median value of 3 meta-analyses (but including several models per meta-analysis), buffers 150 and 220 ft wide are needed for 90 and 95% removal of nitrogen, respectively. According to the meta-analysis of Zhang et al. (2010), buffer widths of 50 ft and at least 100 ft are needed to remove 90% of pesticides and 95% of phosphorus from runoff, respectively. Buffer widths needed to achieve a given level of pollutant removal generally increase as topographic slope increases.

Only 1 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.

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[15] Meta-analysis refers to various approaches for drawing statistical inferences from many separate but similar experiments (Mann 1990).

7.4.6 *Alluvial Water Storage*

Alluvial water storage, also known as bank storage, is a potential 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, USFS, pers. comm.)

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 comprises 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, which equals the distance from the stream bottom to the bank top (Whiting and Pomerants 1997, Chen and Chen 2003).

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 was significant, accounting for 37 to 42% of the annual base flow.

Hydrological studies in the Yakima River basin (Synder and Stanford 2001, and studies cited therein) suggest 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 that suggest 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 ranges from sedges to tall trees (i.e., cottonwood). 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 usually occurs within a site-potential tree height (FEMAT 1993), and therefore, to provide full ecological function, the width of riparian management zones (RMZs) should be one site-potential tree height.[16]

For dryland riparian areas, the width of RMZs should be based on the shade, wood (large and small), or pollutant removal functions. One of these three functions will determine the width of the zone of influence. For grass, herb, shrub, and small tree vegetation types, the zone of influence based on shade or wood, which depend on vegetation height, will be narrower than the zone of

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[16] 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).

influence based on pollutant removal. Therefore, if pollutant removal, including sediments due to ground disturbance, is a concern at a particular site, then RMZ width should be based on the desired removal efficacy for pollutants created at that site. If, for instance, run-off containing excess nitrogen is a concern and a 95% removal efficacy is desired, then a 220 ft wide buffer may be needed. If pollutant removal is not a concern, then RMZ width should be based on site-potential vegetation height (i.e., trees or shrubs), which should provide maximum shade and wood for aquatic habitats.

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 are 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 badly damaged by human activities, and we 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 would include the capacity to support beaver (e.g., Macfarlane et al. 2017) and the vegetation that could 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 functions. 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 function is political. Knowledge of past habitat loss can influence policy decisions to preserve remaining habitats or to restore habitats where practicable.

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[17] 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.

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CHAPTER 8. WATERSHEDS

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 alternatives.

Thus, the purpose of this chapter is to explore the relationship (and the importance) of watershed-scale management to the achieving the goal of abundant, self-sustaining fish and wildlife populations. It assumes that a broader perspective of watershed-scale influences on aquatic ecosystems will ultimately improve the beneficial effects of protective measures being taken at all scales.

The presentation is divided into four primary sections:

1. A conceptual framework that identifies the key watershed processes and their primary influences by, on, and through 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. An evaluation of (and if so, how) riparian management at the site scale, the emphasis of the other chapters in this document, can best support these functions and improve 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 benefits (or gaps) 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 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, large mammals), 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; Carbonneau et al. 2012). 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 (hyporheic) flow with streamflow (Boulton et al. 1998).

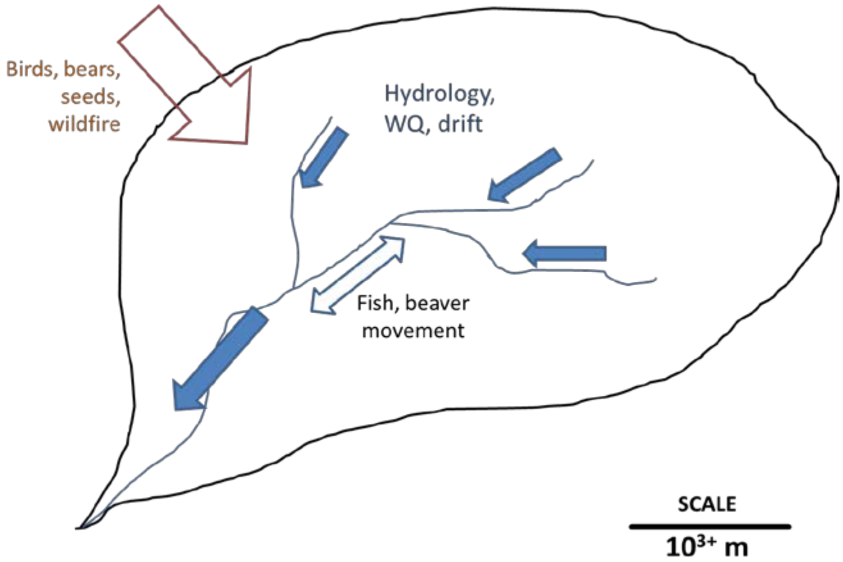


Figure 8.1. Movement of water and biota at the whole-watershed scale.

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, p. 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). 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).

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 east-central part of the image, fragmentation is nonetheless widespread from roads, utility lines, and clearing for residential lots and subdivisions ("f").

At a finer spatial scale, water and biota interact across the semi-permeable and dynamic boundary between the riparian corridor and the watershed uplands (Figure 8.3, left). These areas comprise multiple components: within the riparian corridor, these include alluvial forests, marshes, meadows, in-channel bars and islands, floodplains, surface waters, and woody debris; for the watershed, they might include upland areas, alluvial aquifers, and patches of extensive human modification. Both areas also include the vast variety of organisms that occupy one or more of these habitats.

The components of the riparian corridor itself occupy a particularly dynamic and disturbance-prone region (Figure 8.3, right) that can experience decadal, annual, or even more frequent flood events that locally alter, rearrange, or obliterate those 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" (Ward et al. 2002, p. 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, p. 661). Thus, managers should recognize this dynamic equilibrium as the key attribute of a truly functional riparian–stream system and make the protection of its dynamic *behavior (*rather than any specific *feature*) the overarching goal of riparian management.

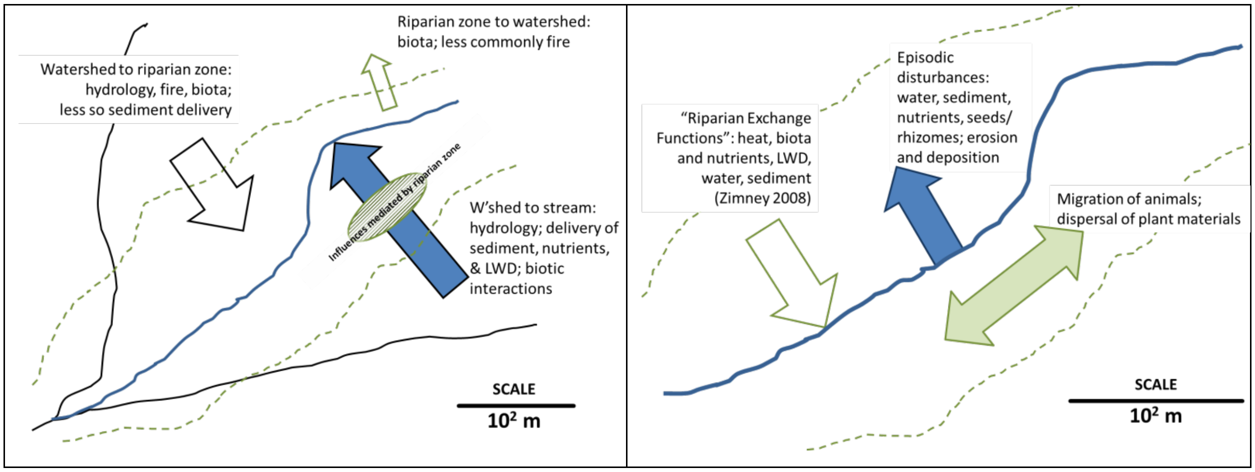


Figure 8.3. Movement of water, biota, and other elements of the aquatic system from the watershed into and through riparian areas (left) and between riparian areas and the stream (right).

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…" (Ryan and Calhoun 2010, p. viii-ix). On the other hand, riparian areas have experienced high levels of disturbance, both directly (e.g., harvest, road building) and indirectly (e.g., windthrow, landslides, and forest fire) from forest management, which also compromise the maintenance of a dynamic equilibrium in riparian-stream systems. Thus, policies applied to forestlands that limit timber management-related disturbance, passively reestablishing currently underrepresented forest seral classes in riparian areas (such as mature forests) with the hope of improving riparian functions, and conserving wildlife species dependent on older forest classes that are less common in upland areas, are all likely necessary components of a successful riparian-management program. However, by themselves they are almost surely not sufficient to achieve desired riparian and instream ecological outcomes.

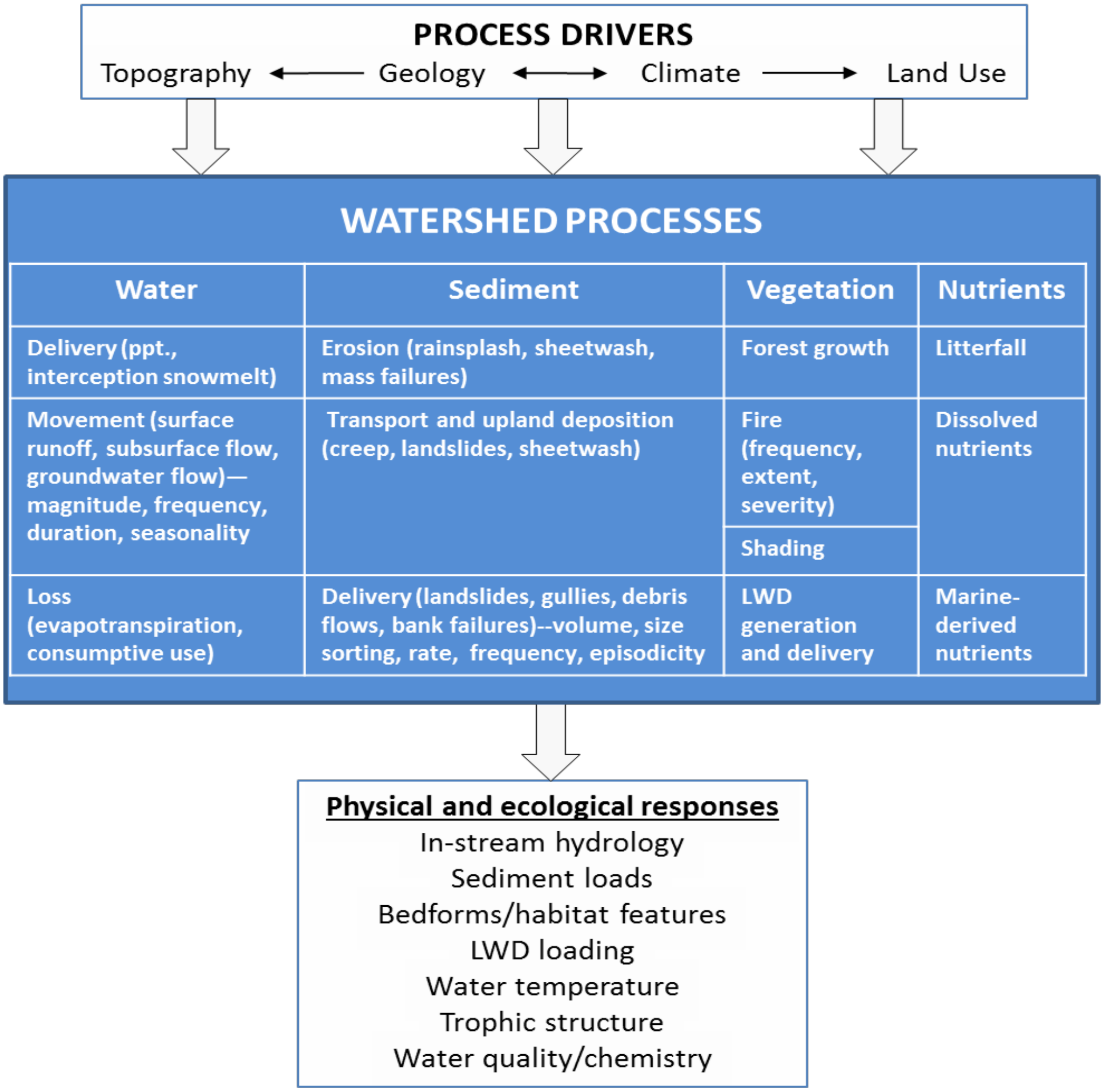
2.1.1 What are "Watershed Processes"?

Although the immediate goal of riparian management is the protection of species and their habitats, "[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" (Frissell et al. 1986, p. 199; 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" (p. 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. 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 Mongomery 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 such 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).

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 beavers (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 historic legacy in the Pacific Northwest (and beyond), the aggregate effects of human activities across whole watersheds can be even more consequential to instream resource, to the extent that "land use," in many settings, becomes no less of a process driver (i.e., a determinant of the flux of energy and materials in a watershed) than is topography or climate.

These concepts are integrated into Figure 8.4, 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, their Figure 1 as an alternative representation), but it sacrifices precision for clarity in acknowledging the overriding importance of this unidirectional perspective. This conceptual model also articulates the range of watershed processes that are embraced within each of the major categories we recognize (water, sediment, vegetation, and nutrients), and it reminds us of the major categories of instream response to the dynamic influence of these watershed processes.



Water	Sediment	Vegetation	Nutrients
Delivery (ppt., interception snowmelt)	Erosion (rainsplash, sheetwash, mass failures)	Forest growth	Litterfall
Movement (surface runoff, subsurface flow, groundwater flow)—magnitude, frequency, duration, seasonality	Transport and upland deposition (creep, landslides, sheetwash)	Fire (frequency, extent, severity) Shading	Dissolved nutrients
Loss (evapotranspiration, consumptive use)	Delivery (landslides, gullies, debris flows, bank failures)--volume, size sorting, rate, frequency, episodicity	LWD generation and delivery	Marine-derived nutrients

Figure 8.4. 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".

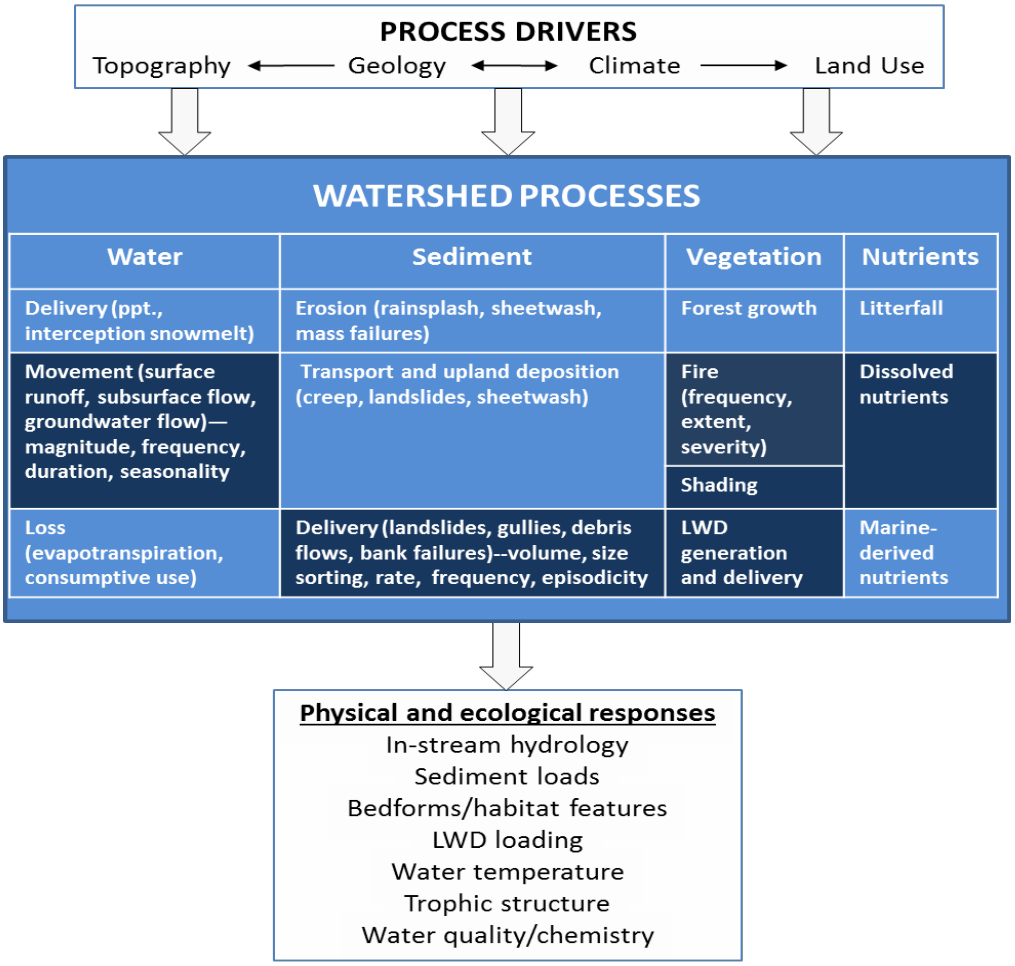


Figure 8.5. The conceptual framework of Figure 8.4, highlighting the subset of watershed processes that most directly affect instream conditions.

8.2.1 *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 (particularly in the Pacific Northwest) large wood. These materials are delivered primarily by surface water flow, subsurface flow, landslides and debris flows (Figure 8.5). 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 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 100 m were associated with disturbances affecting pool retention (cow density, road density) and pool formation (% small trees)" (Anlauf et al. 2011, p. 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, systematic 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" (Allan 2004, p. 257).

Regardless of our present inability to make reliable predictions of instream conditions from watershed attributes, watershed processes do create and maintain instream habitat, and so the determinants of those processes are of indirect, but ultimately of 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.

8.2.2 *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 a variety of potential mechanisms: the loss of forest canopy, which should increase total water yield by reducing interception losses and evapotranspiration; the greater accumulation of snow on canopy-less ground, increasing the total volume of water available during a subsequent rain-on-snow event; and more rapid drainage of surface runoff to stream channels from a newly created road network. The review article of Andréassian (2004) 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" (p. 12). 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 secondary 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 root 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, noting 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 that is commonly expressed throughout the channel network of logged watersheds "could produce a substantial reduction in stored organic matter" (Richardson et al. 2005, p. 930).

8.2.3 *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 potentially transported into streams, 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) 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, 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.

8.2.4 *Urbanization*

Urbanization is widely recognized to be the most intensive land use for 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 energy; 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 set of homogenous, near-impervious surfaces, and the variety of impacts resulting from the human population that arrives shortly thereafter. 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 far-distant watershed 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.

8.2.5 *How does the watershed affect the riparian corridor?*

Changes to the upland watershed and watershed processes can also affect the conditions and functions of the riparian corridor directly (Figure 8.6). 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.

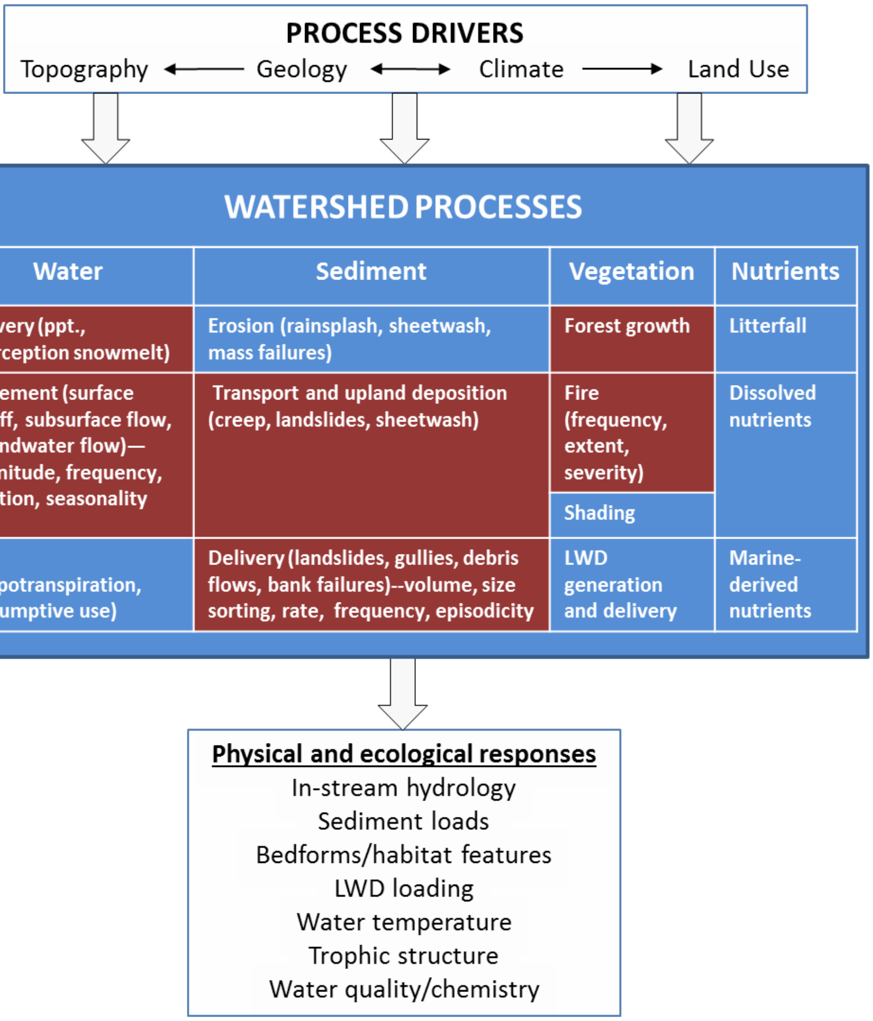


Figure 8.6. The conceptual framework of Figure 8.4, highlighting the subset of watershed processes that can affect conditions in riparian areas.

Upland vegetation "buffers the buffers," providing ecological connectivity between upland plant and animal communities with their riparian counterparts, and providing protection from mechanical disturbances to the riparian areas, such as windthrow. Uplands also constitute a broader environment that either supports or suppresses wildfires that may freely cross the upland/riparian boundary.

Land use impacts are relatively consistent with respect to their direct impacts to the riparian ecosystem itself. All three of the primary types (forestry, agriculture, urbanization) physically remove the native vegetation, with differences only in the degree of impact; the loss of vegetative continuity will consistently affect conditions of biological connectivity and the risks of windthrow and fire regardless of the specific upland land use activity that lies beyond the immediate zone of effects.

The interactions between uplands, riparian areas, and instream conditions are particularly complex with respect to wildfire. Arkle et al. (2010) 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).

8.3 THE ROLE AND LIMITATIONS OF SITE-SCALE RIPARIAN 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, p. 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 the process drivers is not uniform across a watershed, so also the influence of disturbances varies as well. 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 habitat. 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 the various components (Fausch et al. 2002). Adding to this complexity is the fact that habitat is species-specific; what constitutes habitat for one species may be mostly unrelated to what constitute habitat for another aquatic species in the same river.

Thus, if riparian areas are managed with the objective of maintaining "high-quality habitat" for aquatic species, managers must ensure that riparian areas per se contribute what they can (i.e., separate from, but in combination with, watershed processes or drivers) to the long-term maintenance of the complex, 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 extent and replication to support the population, and sufficiently proximate to be accessible from one to another yet sufficiently distal to not be subject to the same deleterious disturbances.

This, then, defines the fundamental challenge of watershed and riparian-zone 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 the habitats that result, 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 MANAGEMENT

"Connectivity" is the term most often used to describe whether and how 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 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, p. 288) reported that "…[spatial] network variables perform better at explaining juvenile coho salmon 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, p. 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, p. 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." (Ward et al. 2002, p. 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 stream/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 3 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.

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 along its longitudinal extent, 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 up- and downstream mobile fish. However, any management outcome that results in "frequent" interruptions to riparian areas, or corridor-impacting crossings for which passage of terrestrial and aquatic organisms is not explicitly facilitated to the greatest degree possible, can be assured to having permanent 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 taken an intentionally expanded perspective, in both space and time, from 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, p. 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.*

This theme is widely appreciated amongst biologists and land managers alike, given the recognition that not every location along a stream is equally influential on the physical behavior or biological integrity. This recognition is complicated, however, by the fact that different perspectives may not assign the same level of "critical 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). A general treatment of the topic such as this cannot resolve the multiplicity of such competing values across the diversity of landscapes in the Pacific Northwest, but this perspective 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 the upland development to the riparian management area, nor to the “porosity” of the boundary between them.

8.5.3 *Patch context matters.*

Riparian management areas surrounding streams within watersheds of different land uses (e.g., forestry, agricultural, urban) must provide some very different functions depending on those adjacent land uses, and typically they do so with widely varying levels of effectiveness. The impacts to a stream from watershed urbanization, for example, span the range from pervasive hydrologic changes (for which buffers are typically irrelevant) to the introduction of exotic species (for which the physical separation 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 debris flows; or potential losses of streambank integrity, habitat complexity, shading and large wood, and allochthonous inputs of organic matter.

“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)” (Hansen et al. 2015, p. 54).

8.5.4 *Connectivity is critical.*

Connectivity in riparian areas occurs not only along-stream (previous section), but also transverse to the flow—from the stream through the riparian area into uplands. Typical riparian management area regulations are generally successful at maintaining connectivity in the along-stream dimension at some minimum width, although the consequences of “limited” interruptions to this longitudinal connectivity (e.g., bridges, utility crossings) are commonly dismissed, a consequence of viewing riparian areas only from the perspective of net area rather than a recognition of the importance of their spatial 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. 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 from upland activities. Continuity of a protected riparian area into uplands that are truly protected as well is widely recognized to provide a range of key habitats for mobile species that depend on interacting with both upland and riparian environments for different activities 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 8.7). 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 #2 above).

Figure 8.7. View of isolated wetland and stream Riparian Management Zones 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 wide.

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” (Jansson et al. 2007, p. 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 a key element of riverine attributes and processes 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).

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, however, these spatial and temporal boundaries bear little resemblance to the true suite of influences that determine instream conditions. In general, 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 habitat. 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 the shade provided by a streamside tree) 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); 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 biologically important habitats, and it suggests that achieving genuine habitat protection (and thus species conservation) is crippled by site-by-site management that is limited to 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 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. 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. The natural frequencies, magnitudes, and durations of natural disturbances (flood and fire being the most common) need to be better understood and then maintained to the greatest extent that surrounding land uses can tolerate—habitats have not only spatial but also temporal dimensions to their creation and support, and they cannot retain their functions if they remain static.
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 that outcome:

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 this 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 toxicants 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; and from even beyond those watershed 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 communities of organisms (Ryan and Calhoun 2010). This will require approaches that allow sufficient space and longevity 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.*

The aggregate presence (or absence) of localized site-scale management actions will influence the overall effectiveness of those actions—the whole can be greater, or less, than the sum of its parts. Particularly in rivers, the intrinsic upstream/downstream connectivity of individual locations is a defining feature of these systems and their biota, and so maintaining this connectivity is an essential component of aquatic ecosystem protection. Although individual site actions can support this goal, they cannot assure its achievement. We increase our chances with the collective actions across all riparian sites, and even more so by considering the entire riverine landscape as the focus of conservation efforts. However, such a chain is only as strong as its weakest link, and so even a single ill-executed treatment 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, however, 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, p. 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 nearby riparian ecosystem.

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CHAPTER 9. FROM SCIENCE SYNTHESIS TO MANAGEMENT IMPLICATIONS

Timothy Quinn and Kirk Krueger

9.1 INTRODUCTION

This chapter summarizes important scientific findings from chapter 1-8 and provides a brief discussion of scientific themes that can inform thoughtful approaches to protecting aquatic systems. The state of the science, as reviewed in Chapters 1-8, supports earlier studies that demonstrate the importance of an intact riparian ecosystem to protect and maintain aquatic systems. Riparian ecosystems are a priority habitat because their composition, structure, and functions dramatically affect their ability to support complex food webs that include fish, invertebrates, amphibians, plants, bacteria, fungi, reptiles, birds, and mammals (Cummins 1974; Johnson and Carothers 1982). Although riparian ecosystems are a small portion of the landscape, approximately 85% of Washington's wildlife species use them (Thomas et al. 1979). Conserving (protecting and restoring) high function to this relatively small portion of the landscape can disproportionally benefit many species and other important goods and services (e.g., Clean Water Act, Endangered Species Act, and flood damage control (NRC 2002).

Our inclusive definition of riparian ecosystems is integral to the development of management recommendations. Riparian ecosystems include areas through which surface and subsurface hydrology connect waterbodies with 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). Following Gregory et al. (1991) and Naiman and Bilby (1998) we include active channels, active floodplains and terraces, and adjacent uplands that contribute organic matter to active channels or floodplains as part of the riparian ecosystem. Riparian areas are transitional (ecotones) between terrestrial and aquatic ecosystems and are distinguished by gradients in biophysical conditions and ecological processes (i.e., rates change), and unique, abundant and diverse biota.

9.2 KEY FINDINGS FROM CHAPTERS 1-8

1. 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-related economic activity (Naiman and Bilby 2001; NRC 2002; Richardson et al 2012), and c) they respond to and interact with watershed-scale processes to help create and maintain aquatic habitats and the other ecosystem services.

2. Stream riparian ecosystems include the riparian zone, active floodplain including riverine wetlands, and terraces, and the adjacent uplands that contribute matter and energy to the

active channel or floodplain (Gregory et al. 1991; Naiman and Bilby 1998) called the zone of influence.

3. The width of the riparian ecosystem is estimated by one Site-Potential Tree Height (SPTH) measured from the edge of the channel, channel migration zone or active floodplain; it also includes wetlands and steep slopes associated with this area. Protecting functions within at least one SPTH is a scientifically supported approach if the goal is to protect and maintain high function of the riparian ecosystem.
4. A near consensus of scientific opinion holds that the most effective and reliable means of maintaining viable self-sustaining fish, especially salmon, and wildlife populations 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 based in part on the complexity of processes that affect the expression of habitat over time and space.
5. The protection and restoration of the 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 protections.
6. Riparian areas and surrounding watersheds are complex and dynamic systems comprised of many interacting components. These interactions across the watershed and through time create the mosaic of conditions necessary for self-sustaining populations of fish, especially salmon, and other aquatic organisms.
7. Impending changes to aquatic systems as a result of climate change increases risk to species already threatened by human activities. The warming effects of climate change on rivers and streams threaten to drastically reduce fish distribution and viability throughout the Pacific Northwest (Beechie et al. 2013).
8. 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 watershed and riparian systems.

9.3 IMPLICATIONS FOR MANAGEMENT APPROACHES

Riparian ecosystems are a traditional focus of conservation efforts because they help mediate the translation of watershed (process) drivers to instream physical, chemical and ecological responses and ultimately to the creation and maintenance aquatic habitat. In other words, riparian ecosystem conditions at a site are governed by hierarchical regional, watershed and reach scale processes. These processes control the hydrologic and sediment regimes, floodplain and aquatic habitat dynamics, and riparian and aquatic biota (Figure 9.2). For example, variable flows of water, driven by local storms, affect wood and sediment recruitment to the stream channel and movement that result in stream channel changes which, in turn, interact with riparian vegetation. Riparian vegetation changes as streambanks and floodplain surfaces erode and aggrade; and as vegetation matures and succession takes place (Stanford et al. 1996; Independent Scientific Group 1999).

Such spatial and temporal heterogeneity suggests that management should maintain a dynamic matrix of habitat conditions that support species at each life stage at large spatial extents (e.g., watershed) and especially to maintain the suite of ecosystem elements responsible for creating and maintaining those conditions. In addition to riparian ecosystems, the watershed is an appropriate scale for management and will require consideration of anthropogenic changes to hydrology, water quality and watershed connectivity (e.g., movement of water, sediment, wood, nutrients, and species).

Anthropogenic changes to rivers and streams have altered aquatic ecosystem composition, structure and functions (Wohl 2005). Human activities affect the key attributes of processes, commonly at the watershed scale (e.g., increasing/decreasing frequency, magnitude, and duration of flood events) as well as directly altering riparian ecosystem composition and structure (e.g., armoring streambanks, channelizing streams, installing levees, removing riparian vegetation, introducing exotic species). Importantly, biological system resistance, i.e., the ability of an ecosystem (e.g., individual species, communities, and food webs) to maintain its current state (Walker et al. 2004), is in part related to its evolutionary history. Landres et al. (1999) suggest that historical conditions provide 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 and provides a meaningful benchmark for assessing potential loss of riparian function and potential risks to fish and wildlife populations (Morgan et al. 1994; Landres et al. 1999). Because habitat heterogeneity is driven by disturbances and recovery processes, Landres et al. (1999) also suggest that maintaining ecosystem functions requires maintaining, or at least emulating, the disturbance processes that create habitat heterogeneity. In other words, the historical template describes the relationships between species and habitats and thus provides context and guidance for managing ecological systems in the future.

A near consensus of scientific opinion holds that the most effective and reliable means of maintaining viable self-sustaining fish, especially salmon, and wildlife populations is to maintain/restore ecosystems to conditions that resemble or emulate their historical range of natural variability (hereafter historical conditions; Swanson et al. 1994; Reeves et al. 1995; Bisson et al. 2009). The scientific rationale for this opinion is that fish and wildlife species have over many millennia adapted to particular disturbance regimes and habitat conditions, and that dramatic deviation from those conditions put species at risk of extinction (Swanson et al. 1994). "Emulation" also supports the appropriate use of engineered or novel management techniques (e.g., stormwater basins to reduce peak flows, placing large wood in streams). Bisson et al. (2009) promote this concept for conserving freshwater habitats of anadromous salmonids.

Further, emulating historical conditions is a holistic as opposed to a reductionist approach that can lessen both the scope and negative consequences of scientific uncertainty associated with other management approaches and climate change. For example, managing a site, reach, or watershed to more closely resemble historical conditions likely encompasses the numerous needs of multiple species and may be a more effective conservation strategy than managing for each species individually (Hunter et al. 1988; Swanson et al. 2004). Maintaining and restoring self-sustaining fish and wildlife populations is an important goal, and WDFW is confident that managing riparian ecosystems towards conditions that emulate historical conditions while addressing watershed-scale processes is the most reliable approach. It may also prove to be less expensive to manage for historical conditions than restoring conditions after impacts (Landres et al. 1999).

FEMAT's (1993) curves are useful conceptual models describing important riparian functions under historical conditions and how those functions commonly change with distance from the stream channel or floodplain (Figure 9.1). Originally, the curves were meant to convey two important points: 1) most of the four riparian ecosystem functions or processes occur within one SPTH, and 2) the marginal return for each function or process decreases as distance from the stream channel increases. Importantly, FEMAT curves ignore site-specific differences in riparian function among stream reaches. In the absence of site-specific information, SPTH have been used to describe the width of the riparian ecosystems in a number of important management settings**.** Protecting functions within at least one SPTH is a scientifically supported approach if the goal is to protect and maintain high function of the riparian ecosystem.

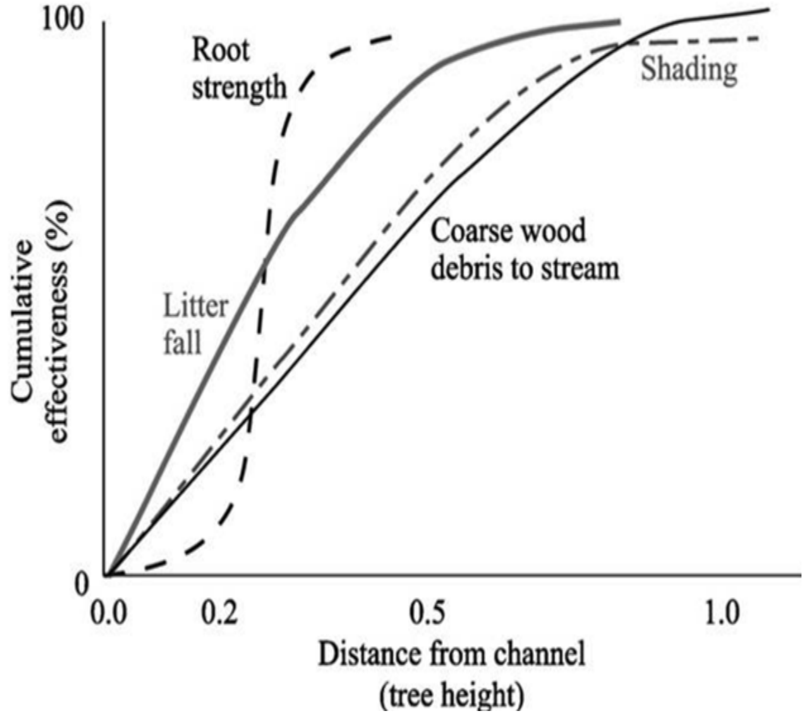

Figure 9.1. The "FEMAT Curves" (FEMAT 1993): generalized conceptual models describing some riparian forest contributions to riparian ecosystem functions and processes as distance from a stream channel or edge of the floodplain increases. "Tree height" refers to average maximum height of the tallest dominant trees (200 years old or greater) and is referred to as site-potential tree height (SPTH).

Even when riparian ecosystems exhibit high ecological integrity, variability at the site-level results in scientific uncertainty regarding the actual shape of function curves. All else being equal, protecting areas less than a SPTH result in lower levels of function, and generally increase risk to aquatic fish and wildlife. The FEMAT model also shows that riparian protection closer to the stream provides a larger number of functions and more of each function per unit width than areas farther from the stream.

Although not all riparian functions are represented by FEMAT curves, large wood recruitment (McDade et al. 1990), litter fall (aka allochthonous nutrient inputs; Bilby and Heffner 2016), and stream shading (Moore et al. 2005) are strongly associated with tree height. 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 (SPTH) has often been used to define the extent of the "riparian ecosystem." However, FEMAT curves dealt strictly with functions associated with vegetation and thus did not attempt to represent all elements of the riparian ecosystem important riparian function or to aquatic species and their habitat. For example, FEMAT curves did not explicitly include areas (i.e., floodplains and stream associated wetlands) to accommodate waters and sediment associated with high flows, or properties of soil that influence instream and streambank sediment dynamics, overland and within-soil water movement (see Chapter 2), nutrient dynamics (see Chapter 6), or pollutant removal (see Chapter 5), and groundwater recharge among others. We believe that FEMAT, which was developed for federal forestlands, did not deal with these ecosystem components because they assumed that protecting the area within a SPTH would necessarily protect most of these elements. Given its utility, the height of site-potential trees has been described for a wide variety of forest types and can be readily found in riparian management literature. Fox (2003) found that mean heights of dominant trees in riparian old-growth forest of Washington range from 100 to 240 feet. 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 2015). Site productivity is described quantitatively through a *site index*, which is the average height, that dominant trees of a particular species are expected to obtain 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 and can be found in Appendix 1.

FEMAT (1993, p. 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 at a substantial rate for more than 200 years (Burns and Honkala 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.

9.3.1 *Complexity*

System complexity is a common unifying topic of the science chapters. Riparian ecosystems and their watersheds are dynamic systems comprised of many interacting components. These interactions are sometimes nonlinear and often play out at different rates or frequencies. A dynamic system means that the behavior of the system at a point in time influences the behavior of the system in the future, and nonlinear interactions means that a small change in one component can have a large effect on another system component. Predicting the effects of management actions in dynamic systems can be very challenging.

Frissell et al. (1986) helped conceptualize watersheds as hierarchical systems where larger scale components, structures, and processes acts primarily in the downstream direction on smaller scale components, structures, and processes (Figure 8.1). Factors that affect processes 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 function of watersheds, streams and riparian ecosystems do not simply arise from these drivers. They are created and modified by continuous and episodic, and predictable and unpredictable fluxes of materials (particularly water, sediment, and wood) and energy, which are widely termed watershed processes. Through time, dynamic mosaics of riparian and aquatic habitat conditions form, are maintained, and change through space in predictable (e.g., seasonally) and unpredictable ways (e.g., due to random events like storms, fires, and landslides).

These types of dynamics, are intrinsic properties of all ecosystems and important to the creation and maintenance of fish and wildlife habitat that individual fish, populations, and communities, including salmon (Merz et al. 2015), depend on (Fausch et al. 2002; Ward et al. 2002; Wiens 2002). The availability of multiple types of habitat (i.e., habitat heterogeneity) within the same stream and among streams may be essential for the persistence of multiple life history strategies in salmon species (Bisson et al. 2009; Hilborn et al. 2003; Merz et al. 2015; Waples et al. 2009) and the coexistence of populations of other aquatic and riparian species (Bellmore et al. 2015).

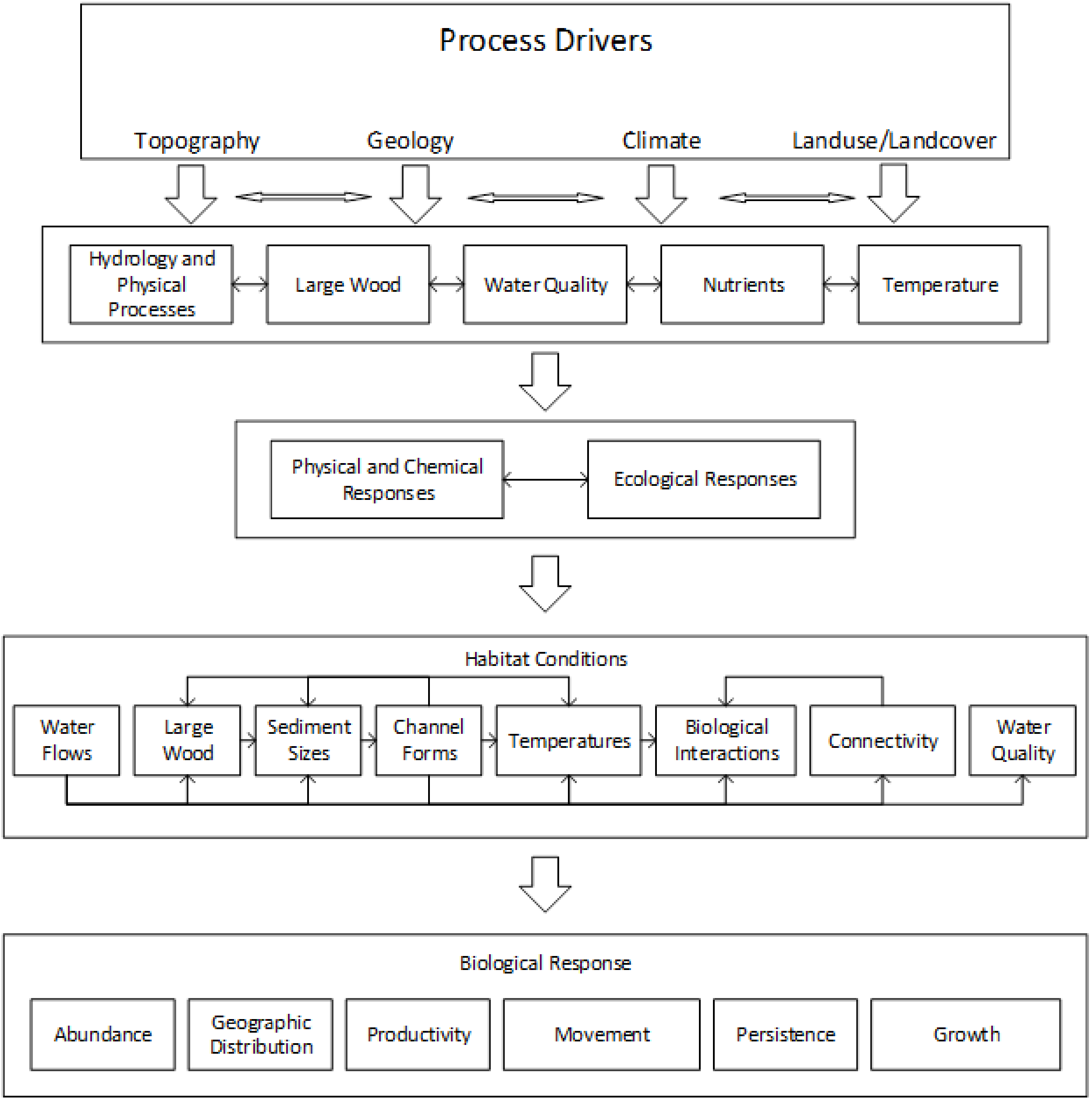

Figure 9.2. Conceptual watershed model describing some relations between process drivers on functions, largely mediated by riparian ecosystem structure and composition, resulting in a set of physical, chemical and ecological conditions that ultimately compose habitat for aquatic species. Habitat quality and quantity, measured on a species- or life history stage-specific basis, can be characterized by elements of biological response. (See Chapter 8 for more detail).

9.3.2 *Connectivity*

Many chapters in this volume discuss the related themes of longitudinal, lateral, and horizontal connectivity. Connectivity is vital to aquatic ecosystem function. Streams and their associated riparian ecosystems transport water, wood, sediment, nutrients, and species predominantly, but not exclusively in the downstream direction. Longitudinal (upstream to downstream) fragmentation of streams, for example by road crossings not designed to accommodate passage of fish, sediment and wood, can change aquatic species composition and decrease species richness by affecting movements of fish and other animals. Water storage reservoirs and undersized culverts reduce the transport of large wood and sediment from upstream areas. Loss of lateral and horizontal connectivity between streams and their floodplains due to levees, flow management, and stream degradation can alter processes such as water storage, hyporheic exchange, and nutrient cycling. Maintaining connectivity in these ecosystems likely requires management consideration at local and watershed extents over the long term.

The importance of connectivity is also supported by our improved 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., water temperatures) is increasingly acknowledged as important to fish population persistence, particularly in light of climate change. Therefore, management should ensure that diverse, suitable habitat conditions are available and accessible at the spatial extents at which aquatic species use them. Because the processes that affect habitat conditions, especially at large spatial extents, often play out over long time periods (e.g., sediment transport and deposition and large wood growth and recruitment), management must account for rates of both disturbance and recovery. What appeared to be poor spawning habitat for salmon in 1920 may be important spawning habitat in 2020. Maintaining connectivity at ecologically important scales requires that rates of recovery of riparian ecosystem function and connectivity across the watersheds equal or exceed natural and anthropogenic disturbances that impair functions and connectivity.

9.3.3 *Causal Relations and Uncertainty*

Effective use of scientific information requires understanding its uncertainty (Fischhoff and Davis 2014), so, we provide a more complete discussion here than was possible in individual chapters. There are many sources of, and frameworks for categorizing, uncertainty (e.g., Hilborn 1987; Suter et al. 1987; and Wynne 1992). We find the classification framework described in Stirling and Gee (2003) to be useful in describing scientific uncertainty, particularly uncertainty associated with the use of predictive models. Management of riparian areas and watershed are based on predictive models that relate particular causes (management actions) to result (biological outcomes). For the sake of clarity, we use the effect of vegetation management on stream temperature as case study. The best predictive model is one that includes all causal factors that can affect outcomes and provides high predication ability

Four classes of scientific uncertainty can be considered (Stirling and Gee 2003) when making predictions using models. Two of those four classes; Class 1 uncertainty (where we have sound mechanistic understanding of cause and effect and high prediction ability), and Class 4 uncertainty (where we have poor mechanistic understanding and low prediction ability), are rarely encountered in riparian ecosystem management and we do not talk about these further

Class 2 and Class 3 uncertainty are more relevant to riparian management. Class 2 uncertainty (poor mechanistic understanding and high prediction ability) can be achieved using correlative statistical (hereafter statistical) models, but prediction to situations with conditions outside the range of those used to develop the models are highly uncertain. Much of the scientific literature includes statistical models based on correlations and those models are used by managers as predictive tools, often without careful consideration of uncertainty. We suggest that this is relatively uncommon because we usually select statistical predictors based on our mechanistic knowledge.

Class 3 uncertainty (sound mechanistic understanding and low prediction accuracy) is the uncertainty most commonly faced in riparian ecosystem management. That is, we can identify the important casual mechanism (components and structures) and often make confident prediction of the direction of the effects on management on outcomes, but cannot make accurate and precise predictions.

For example, the studies presented in Figure 9.2 represent statistical models relating changes in riparian zone (buffer) width to changes in stream temperature. Data in Figure 9.2 are collected by measuring stream temperature before and after a reduction in the width of a riparian forest buffer which in turn decreases shade to the stream. Model output (prediction) is change in temperature related to specific buffer widths. We know with high certainty that many of the important mechanisms describing the relationship between riparian functions and riparian management actions (e.g., FEMAT). For instance, we know that shade can be a major determinant of summer stream temperatures and thus is commonly used in statistical models. Further, we can reliably predict how management actions in riparian ecosystems will *generally* change riparian functions. However, because of the uniqueness of each site (location in watershed, reach type, disturbance history, etc.) predicting the exact change in temperature from a specific amount of shade removal comes with a high level of uncertainty. Thus, our predictive models tell us that reducing shade usually increases stream temperatures, but they cannot predict *exactly* how much shade reduction will result in a specific temperature increase. For example, a manager who attempts to use the statistical model in Figure 9.2 to predict how a reduction from a fully forested 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°, and for another study between -0.5 to nearly 5 C°. Uncertainty in this case describes our inability to accurately predict exact outcomes. Similarly, a statistical model for wood recruitment (Figure 9.4) is consistent with the idea that large wood recruitment generally increases with distance from the stream, but exactly how much wood is recruited within a specific distance from a stream varies widely depending on site-specific characteristics and stochastic processes.

These types of statistical models used in natural resource settings are usually informed by a small numbers of non-representative studies. Typically, few studies have been completed within any region (e.g., ecoregion), and thus most statistical models limited in their applicability to new areas and in their predictive reliability (i.e. class 3 uncertainty). Non-representative sampling in a study occurs when researchers (wisely) choose study sites that purposely exclude extraneous factors that can add unwanted variation to the independent variable(s) of interest. Because of these issues, the mean results of statistical models do not consistently predict the precise magnitude of management effects at any specific location.

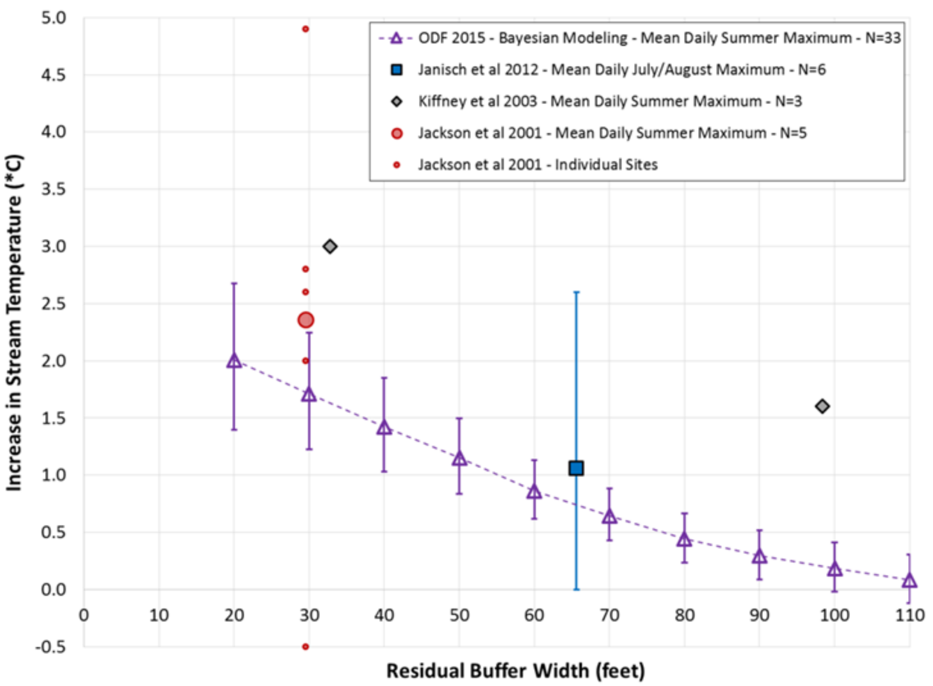

Figure 9.3. Observed temperature response associated with "no-cut" riparian buffers with adjacent clearcut harvest. Only studies that employed a Before-After-Control-Impact design and conducted in Pacific Northwest forests are included. Bayesian modeling results (and 90% credible intervals) were derived from data collected as part of Groom et al. 2011. Analyses provided by P. Leinenbach, USEPA Region 10) (See Chapter 4 for a more complete description of this figure).

Understanding mechanistic models can provide insights into statistical model imprecision and class 3 uncertainty. Unlike statistical models, mechanistic models can capture all important relationships among causal variables and an outcome. That is, if we measure all causal variables (Figure 9.3) including shade from riparian vegetation, we can more precisely and reliably predict stream temperature at the site scale than can a statistical model. Management effects on stream temperature can be generally understood, but not precisely and reliably predicted by most statistical models, because stream temperature is controlled by many variables, including riparian shade. Additionally, other functions described in the temperature model as shown in Figure 8.3, can affect stream temperatures (and often each other).

Mechanistic models have the potential to provide precise and site-specific information however, the cost of data collection has made them largely impractical for managers. Moreover, because the assumption that functions remain constant is rarely met in these dynamic systems, the applicability of mechanistic models is also limited unless models are continually updated – a condition that is true of statistical models as well. We provide this comparison between models types so that managers understand that while statistical models reliably tell us the direction of change, they should only be used to answer site-specific questions with a full understanding of the level of precision and uncertainty they provide.

For the time being, statistical models will likely continue to be an important source of information to managers. With this in mind, we should strive to improve these tools – preferably within an adaptive management framework. The most obvious approach to improving statistical models is to address the known shortcomings listed above, i.e., strive to understand cause (human actions) and effect mechanism (undesirable changes the environment); conduct more studies stratified by important variables (e.g., variable known to affect the function of interest) across a broader range of physical conditions where those models are expected to be used; eliminate incentives to conduct studies that include non-representative sampling; and increase the use of multivariate approaches to statistical models, e.g., including causal predictors, known to via mechanistic models in study designs.

Improvement of statistical models may reduce, but will not eliminate uncertainty. Our challenge is to identify and effectively minimize uncertainty and to make management decisions that are appropriate given our uncertainty and the associated risk.

Scientific uncertainty can represent risk and thus increase the difficulty of policy decisions. Risk is composed of two parts, the magnitude of an adverse impact and the probability of that adverse impact occurring. An adverse impact could be conceived as over- or under-estimating the level of protection needed to meet a full riparian function. Unexpected adverse impacts associated with under-protections are often expensive and time consuming to fix even with the presence of an adaptive management system, e.g., the harvest of a 400 years old forest, or locating cities in flood plains or the estuary of a major river.

Risk associated with scientific uncertainty is often addressed in two ways: by applying the precautionary principle, that is, erring on the side of caution particularly when dealing with moderate to high probabilities of impacts that are difficult to undo (large magnitude); and through the thoughtful application of adaptive management.

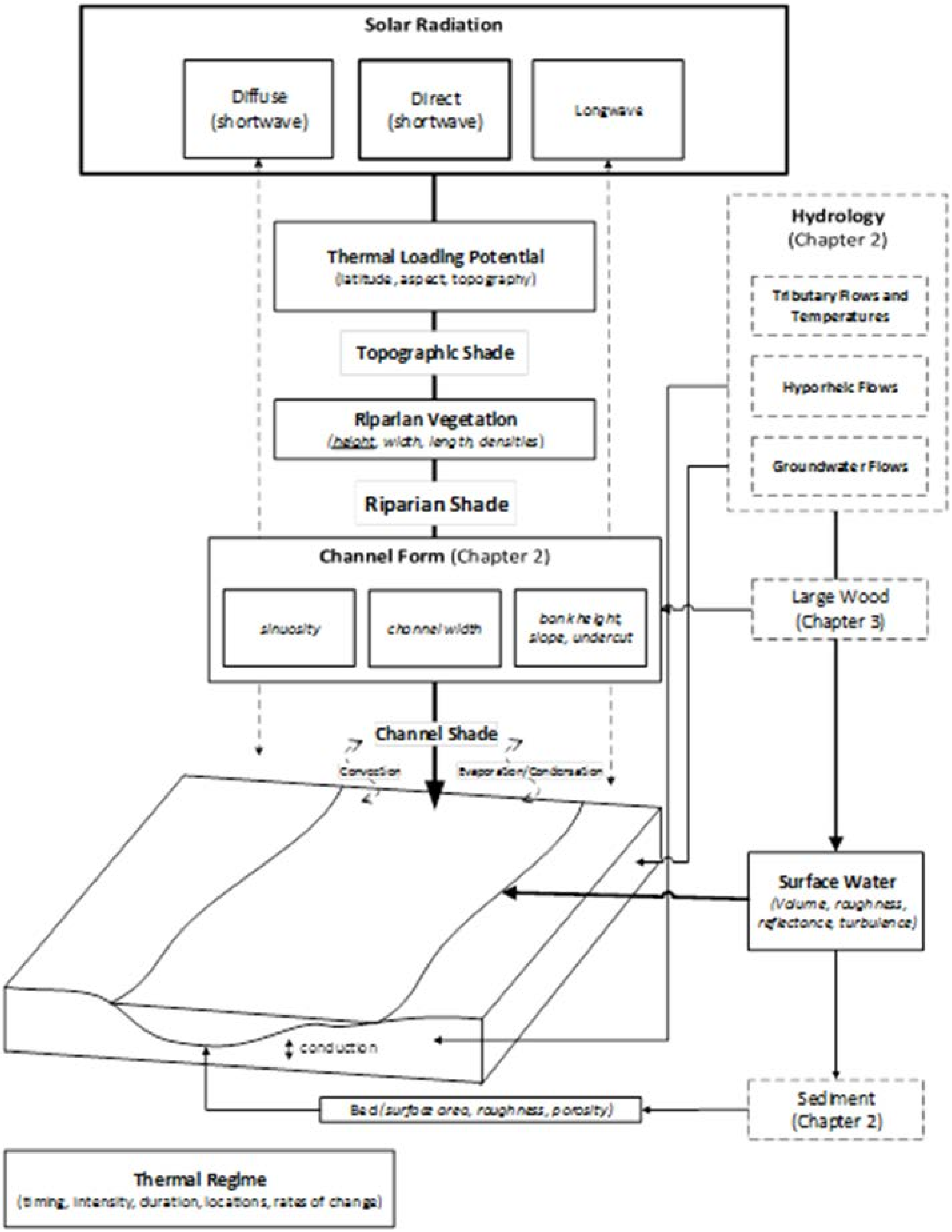

Figure 9.4. Conceptual model of some factors that affect the thermal regime of streams. Relative importance is noted by line weight and arrows indicate directions of flows. Note that most of the elements in the model are influenced by human activities and that many components and structures are not presented here. See Chapter 4 for more detail.

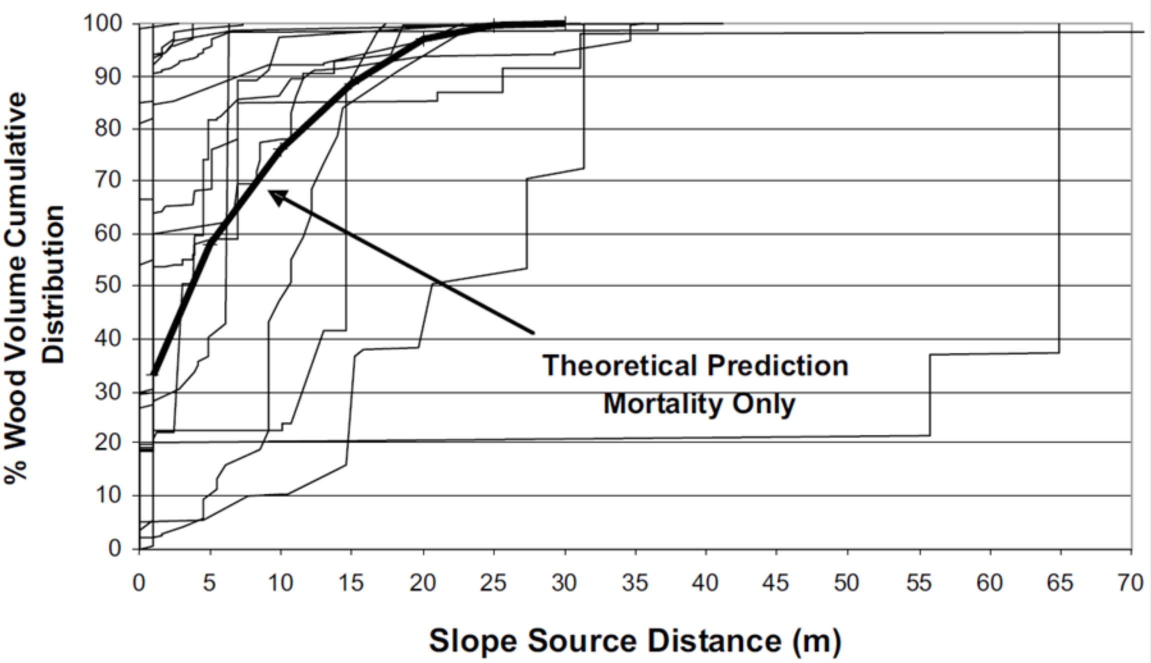

Figure 9.5. Sixteen large wood recruitment curves for second-growth redwood forest on 16 sites in northern California (from Benda et al. 2002). The theoretical prediction curve is based on a mortality recruitment only using random 360° fall trajectories. Curves to the left of the theoretical curve are 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.

9.4 CLIMATE CHANGE

Impending changes to aquatic systems as a result of climate change increases risk to species already threatened by human activities and we expect climate change to adversely affect the performance of models, especially statistical models, which predict the effects of management actions on riparian ecosystem functions. The warming effects of climate change on rivers and streams threaten to drastically reduce fish distribution and viability throughout the Pacific Northwest (Beechie et al. 2013). Human associated CO_2 emission and alterations to the landscape influence water temperature by changing factors that regulate stream temperature, including climatic drivers (e.g., air temperature and precipitation), discharge, stream morphology, groundwater interactions, and riparian canopy (Poole and Berman 2001). The Pacific Northwest has experienced a significant warming trend in summer stream temperatures of approximately 0.22 C/decade between 1980 and 2009 (Isaak et al. 2012). August stream temperatures are projected to increase an average of 2.83 C by the 2080s (Isaak et al. 2015) and stream temperatures as expected to increase in all seasons of the year.

In addition to climate impacts and as noted in Chapter 4, increases in water temperature can result from decreased streamflow, simplification of stream channels (e.g., increased width-to-depth ratio and reduced hyporheic and groundwater exchange), and reduction of riparian vegetation cover (Poole and Berman 2001). These modifications are often the consequence of land use activities such as riparian vegetation removal (Beschta et al. 1987; Moore et al. 2005), water diversions, unmanaged livestock grazing (Kauffman and Krueger 1984; Belsky et al. 1999), and stream channelization associated with roads and levees and other forms of human development (Simon and Rinaldi, 2006). Water temperature is widely recognized as one of the most important environmental factors influencing the geographic distribution, growth, and survival of fish and other aquatic organisms (Regier et al. 1990; Armour 1991; McCullough 1999).

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APPENDIX 1: CATEGORIZATION OF INFORMATION SOURCES

A1.1 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 WDFW;
(iii) External peer review: Review by persons that are external to and selected by WDFW;
(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 WDFW employees or other individuals; or
(viii) Other: Sources of information that do not fit into categories i - vii.

A1.2 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 and the US National Research Council (NRC); symposia volumes sponsored by professional organizations (e.g., American Fisheries Society); and graduate theses and dissertations to category i – independent peer review. Federal documents including those published by the Fish and Wildlife Service (USFWS), Environmental Protection Agency (EPA), Geologic Survey (USGS), Forest Service (USFS), National Academy of Engineering, etc., and those published by Canadian Wildlife Service were assigned to category viii. Others references were assigned as described above.

APPENDIX 2: DETERMINING SITE-POTENTIAL TREE HEIGHT

A2.1 INTRODUCTION

Several major contemporary riparian conservation plans/strategies (USDI & UDSA 1994, USDI & UDSA 1995, DNR 1997, DNR 2005) base the width of riparian management zones (RMZs) on site-potential tree height. In forested ecoregions, site-potential tree height is used to define the width of one side of a riparian ecosystems in which some but not necessary all riparian ecological functions are provided (Gregory 1997). FEMAT (1993, p. 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 (sensu Franklin and Spies 1991). This reflects FEMAT's belief that old-growth forest conditions within a site-potential tree height of a stream typically provides for full riparian ecosystem functions. Heights of dominant trees in old-growth riparian forests of Washington range from 100 to 275 feet (Curtis et al. 1974), depending on site class.

The Northwest Forest Plan (USDI & UDSA 1994) based site-potential tree height on the height of trees in an old-growth riparian forest. In contrast, the habitat conservation plan (HCP) for DNR's forested state trust land defined site-potential tree height, as the potential height of trees in a mature riparian forest, where "mature" was defined as 100 years old. Consequently, the site-potential height for DNR's HCP could range from 86 to 215 feet (DNR 1997). The Forests and Fish HCP (DNR 2005) also used 100 years as the site-potential age. The use of 100 as opposed to 200 years of age to define site-potential tree height should be considered part of negotiated agreement among stakeholders at least during for the Forest and Fish Agreement (Timothy Quinn, personal communication). The DNR state trust lands HCP and the Forests and Fish HCP both used the 100-year site-potential tree height for riparian RMZ widths, however, site-potential tree heights for the trust land HCP range from 86 to 215 feet and site-potential tree heights for the Forests and Fish HCP range from 90 to 200 feet. The difference is due to the sources for site index curves. The state trust land HCP used King (1966) and Forests and Fish HCP used McArdle et al. (1961).

RMZ widths of both HCPs were based on the 100-year site-potential tree height, but Douglas-fir, the most common tree species in managed forests of western Washington, may not achieve full height until 400 years or older (McArdle et al. 1961). The site-potential height of a 100-year-old Douglas fir is about 75% that of a 200-year-old Douglas fir (King 1966, Curtis et al. 1974). Full ecological function is achieved with RMZs that are at least as wide as the height of old-growth riparian forest (FEMAT 1993, p. V-27). Therefore, in theory and empirically, RMZ widths based on 100-year site-potential tree height will result in a reduction in some ecological function compared to 200-year site-potential tree height.

A2.2 STEPS FOR DETERMINING SITE-POTENTIAL TREE HEIGHT

The easiest way to determine site-potential tree height for a particular location is through resources on the internet. The Washington Department of Natural Resources provides online interactive maps of site productivity classes for all nonfederal and nontribal forestlands in Washington. To access the map, go to http://www.dnr.wa.gov/ and follow these steps:

1. Click on “Forestry” box.
2. Select “Forest Practices” from menu that appears within Forestry box.
3. Click on “Forest Practices Application Review System (FPARS)”.
4. Click on “Forest Practices Activity Mapping Tool”.
   That will take you to https://fortress.wa.gov/dnr/protectiongis/fpamt/index.html#
5. Click on “Map Themes” (upper left corner of map).
6. From drop down menu select “Site Class”.
7. Zoom into map repeatedly until site class polygons appear (about seven clicks on “+”).
8. Click on “Legend”, near upper left corner of screen.

Use the legend to determine the site class of your location. Use Table A2.1 to determine the 200-year site-potential tree height for your location.

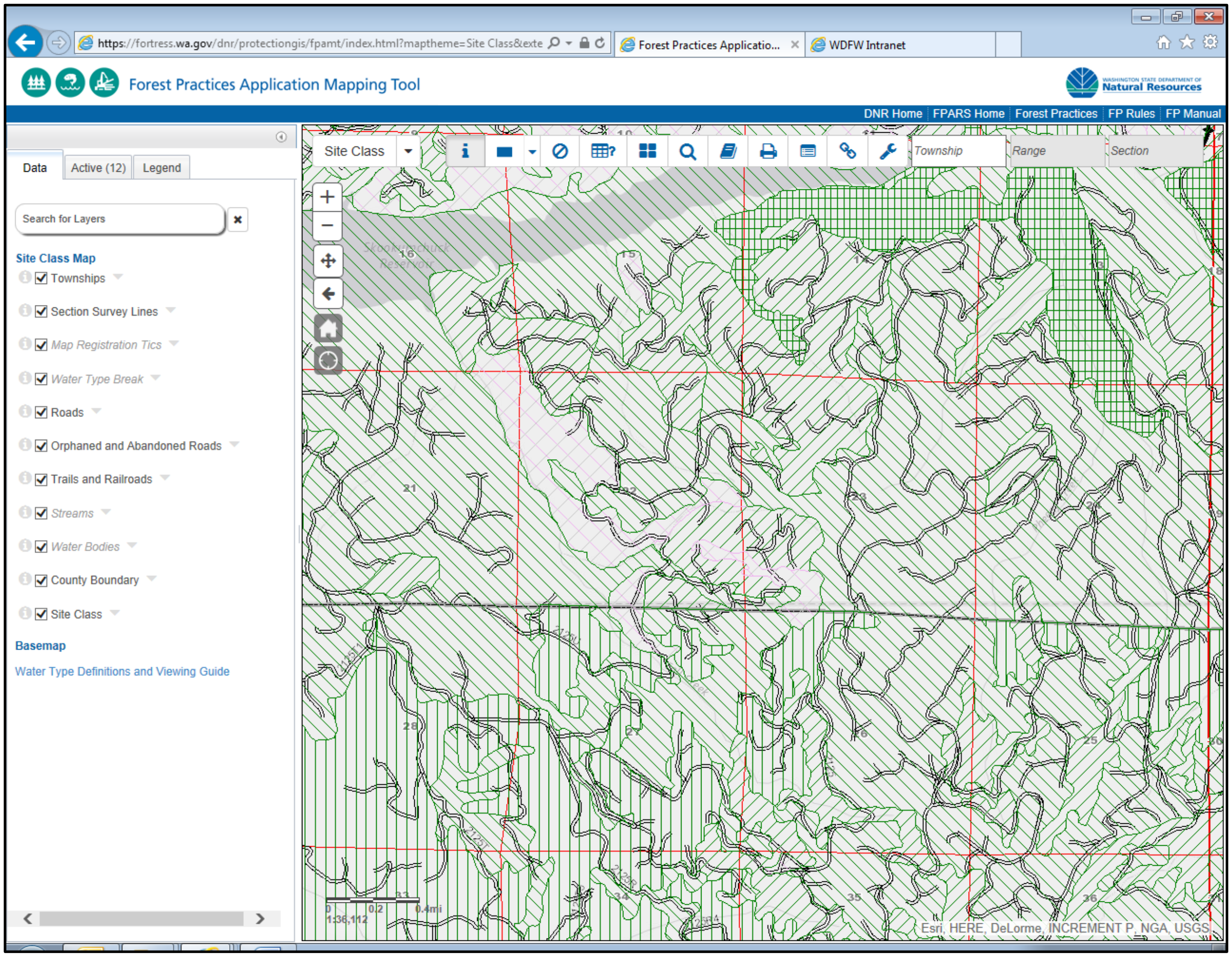
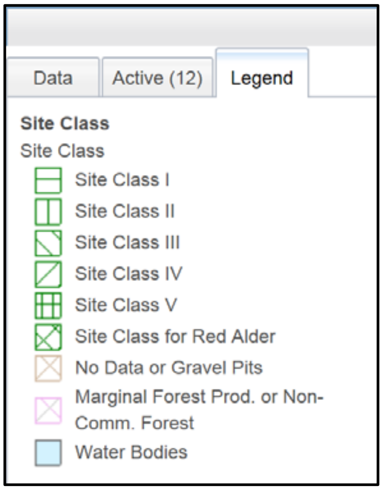


Figure A2.1. Example of interactive on-line site class map. Black lines are roads. Red lines are section boundaries. See Figure A2.2 for site class legend. Map available at https://fortress.wa.gov/dnr/protectiongis/fpamt/index.html#.



Site Class

Site Class
- Site Class I
- Site Class II
- Site Class III
- Site Class IV
- Site Class V
- Site Class for Red Alder
- No Data or Gravel Pits
- Marginal Forest Prod. or Non-Comm. Forest
- Water Bodies

Figure A2.2. Legend for site class map.

Table A2.1. 200-year site-potential tree heights in feet by site productivity class. Two different estimates give approximately the same heights.

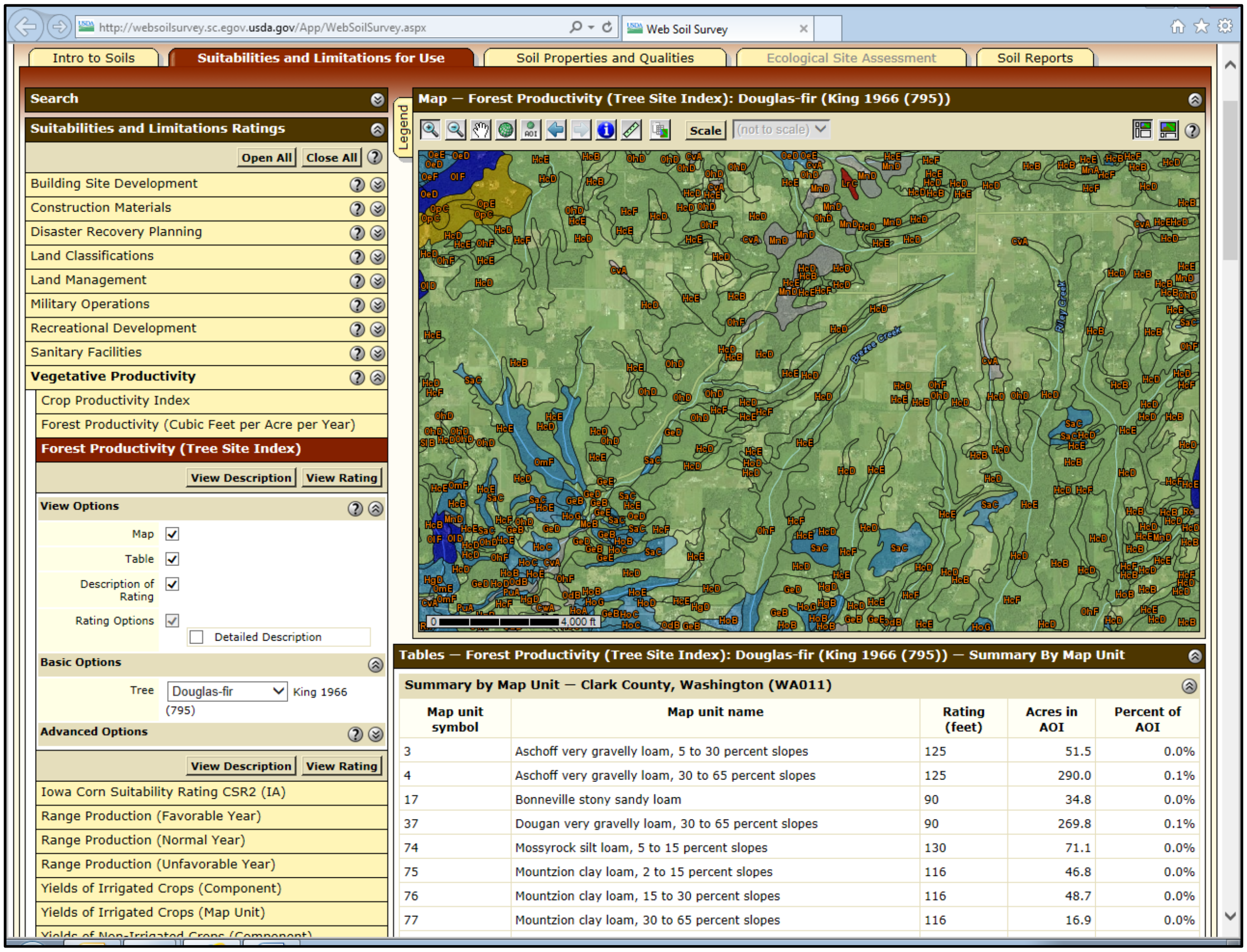
Site Class	King (1966)	Curtis et al. (1974)
I	276	275
II	225	223
III	185	183
IV	146	145
V	100	99

Another source of information on site productivity that can be used to determine site-potential tree height is the Web Soil Survey (WSS) provided by the Natural Resources Conservation Service. To access the map, go to http://websoilsurvey.sc.egov.usda.gov/App/HomePage.htm and follow these steps:

1. Click on the green "WSS" button.
   That will take you to http://websoilsurvey.sc.egov.usda.gov/App/WebSoilSurvey.aspx
   You can use this site to access NRCS soil survey data.
2. Define an area of interest (AOI).
   a. Under "Quick Navigation", click on "Soil Survey Area."
   b. Select state and county from drop down menus.
   c. Select soil survey dataset with radio button.
   d. Check "Show Soil Survey Areas Layer in Map."
   e. Click on "Set AOI."
3. Click on "Soil Data Explorer" tab (above map).
4. If not already selected, click "Suitabilities and Limitations for Use" tab (above map).
5. Click on "Vegetative Productivity" in menu on left.
6. Click on "Forest Productivity (Tree Site Index)."
7. Check "Map", "Table", and "Description of Rating."
8. Select tree species from drop-down menu.

   a. In western Washington, select Douglas-fir (King 1966).
   b. In eastern Washington, select Douglas-fir (Cochran 1979) or Ponderosa Pine (Meyer 1961), depending upon dominant tree species at the site.

9. Click on “View Rating”
   You will end up with a map of all of the soils within the AOI. Alphanumeric labels on the map are the soil number or soil type. Only colored polygons have a site index “rating” for the selected tree species. Scroll down toward bottom of window to view site indices for each soil type.
10. Use map tools (upper left corner of map) to zoom into project area.
    a. Zoom into map repeatedly until soil symbols appear on map.
11. To set a new AOI, click on “Area of Interest (AOI)” tab (upper left corner).
12. Click on “Clear AOI”.
13. Repeat steps 2 through 10.



Map ☑
Table ☑
Description of Rating ☑
Rating Options ☑
☐ Detailed Description
Tree: Douglas-fir King 1966 (795)

Map — Forest Productivity (Tree Site Index): Douglas-fir (King 1966 (795))

Tables — Forest Productivity (Tree Site Index): Douglas-fir (King 1966 (795)) — Summary By Map Unit

Summary by Map Unit — Clark County, Washington (WA011)

Map unit symbol	Map unit name	Rating (feet)	Acres in AOI	Percent of AOI
3	Aschoff very gravelly loam, 5 to 30 percent slopes	125	51.5	0.0%
4	Aschoff very gravelly loam, 30 to 65 percent slopes	125	290.0	0.1%
17	Bonneville stony sandy loam	90	34.8	0.0%
37	Dougan very gravelly loam, 30 to 65 percent slopes	90	269.8	0.1%
74	Mossyrock silt loam, 5 to 15 percent slopes	130	71.1	0.0%
75	Mountzion clay loam, 2 to 15 percent slopes	116	46.8	0.0%
76	Mountzion clay loam, 15 to 30 percent slopes	116	48.7	0.0%
77	Mountzion clay loam, 30 to 65 percent slopes	116	16.9	0.0%

Figure A2.3. Example of interactive on-line soil survey map. Alphanumeric symbols denote various soils types, which are separated by thin black lines on map. The “Rating” column in table below map is the site index for Douglas-fir.

The “Rating” column in table below soils map is the site index for the selected tree species. In western Washington, to translate Douglas-fir site index to site productivity class see Table A2.2, and then use Table A2.1 to determine the 200-year site-potential tree height for your location. In eastern Washington, use Table A2.3 for Douglas-fir or Table A2.4 for Ponderosa pine, depending upon the dominant tree species in the riparian ecosystem. When determining the dominant tree species in the riparian ecosystem, assess the ecosystem's zone of influence, which is typically the upland portion of the riparian ecosystem. Use the floodplain, channel migration zone, or classical riparian zone only when no trees exist in the zone of influence.

Table A2.2. Translation of Douglas-fir site index (King 1966) to site productivity classes. Use Table A2.1 to determine 200-year site-potential tree height for Douglas-fir.

Site Index Range		Site Class
Lower Limit	Upper Limit	
135	160	I
115	134	II
95	114	III
75	94	IV
50	74	V

Table A2.3. Site-potential tree heights in feet by site index for interior (east side) Douglas-fir (Cochran 1979). Interior Douglas-fir can live for over 300 years, however, diameter growth becomes extremely slow and height growth practically ceases after age 200 years (Burns and Honkala 1990). Height equations in Cochran (1979) only valid for stand ages less than 180 to 190 years, depending upon site class. Stand age is total age, which was determined by adjusting breast-height age with equation 10 in Thrower and Goudie (1992).

Site Index	Height at Age 180 to 190 Years
50	88
60	104
70	120
80	135
90	151
100	167
110	182

Table A2.4. Two-hundred year site-potential tree heights in feet by site index for Ponderosa pine (Meyer 1961). Stand age is total age.

Site Index	Height at Age 200 Years
40	50
50	64
60	80
70	97
80	112
90	128
100	143
110	157
120	172
130	187
140	198

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