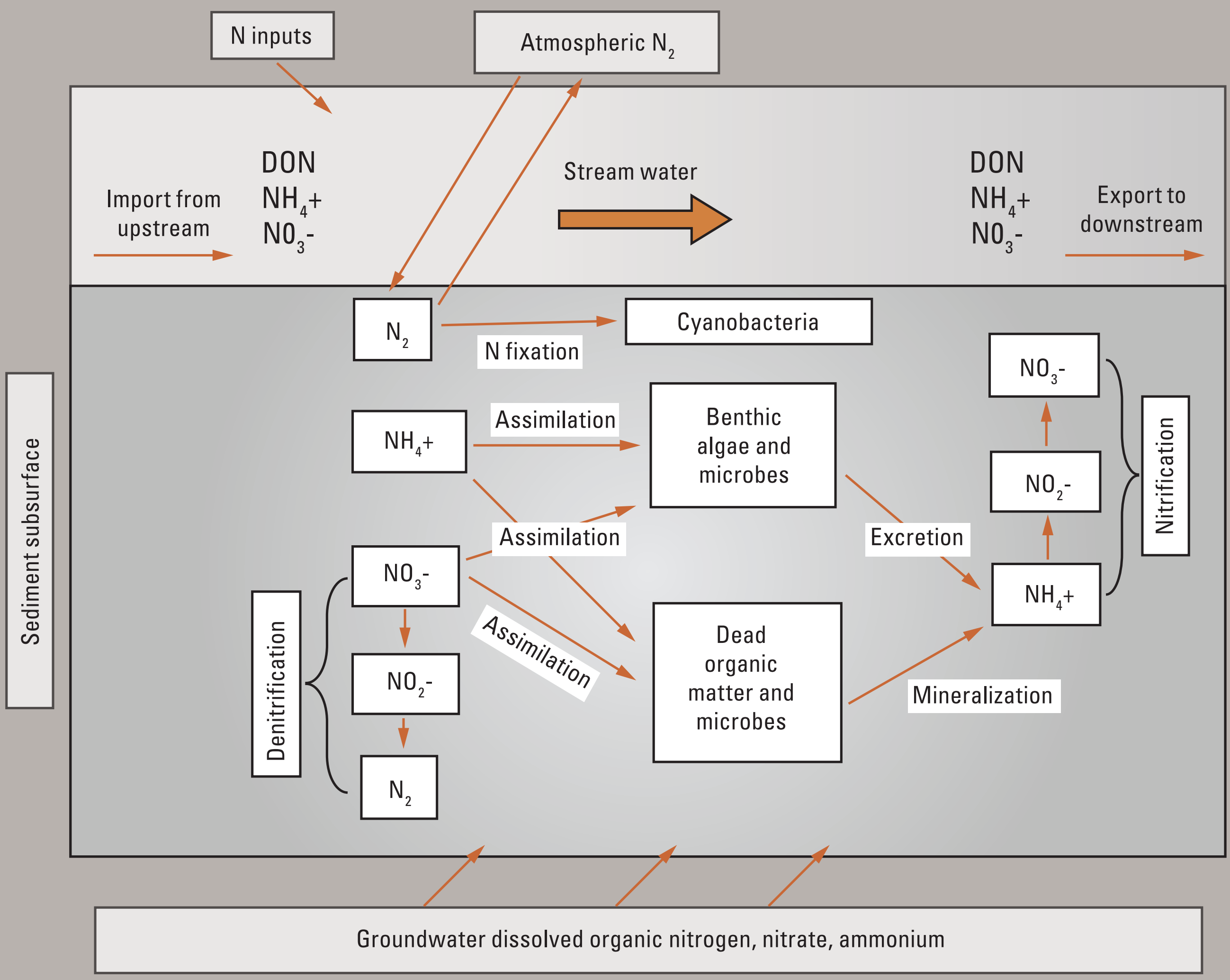
Prepared in cooperation with the Washington State Department of Ecology

Nutrient Attenuation in Rivers and Streams, Puget Sound Basin, Washington



Scientific Investigations Report 2015–5074
Version 1.1, February 2016

U.S. Department of the Interior
U.S. Geological Survey

Nutrient Attenuation in Rivers and Streams, Puget Sound Basin, Washington

By Rich W. Sheibley, Christopher P. Konrad, and Robert W. Black

Prepared in cooperation with the Washington State Department of Ecology

Scientific Investigations Report 2015–5074
Version 1.1, February 2016

U.S. Department of the Interior
U.S. Geological Survey

U.S. Department of the Interior
SALLY JEWELL, Secretary

U.S. Geological Survey
Suzette M. Kimball, Acting Director

U.S. Geological Survey, Reston, Virginia
First release: 2015
Revised: February 2016 (ver. 1.1)

For more information on the USGS—the Federal source for science about the Earth, its natural and living resources, natural hazards, and the environment—visit http://www.usgs.gov or call 1–888–ASK–USGS.

For an overview of USGS information products, including maps, imagery, and publications, visit http://www.usgs.gov/pubprod/.

Suggested citation:
Sheibley, R.W., Konrad, C.P., and Black, R.W., 2015, Nutrient attenuation in rivers and streams, Puget Sound Basin, Washington (ver. 1.1, February 2016): U.S. Geological Survey Scientific Investigations Report 2015–5074, 67 p., http://dx.doi.org/10.3133/sir20155074.

ISSN 2328-0328 (online)

Contents

Figures

Figures

Tables

Conversion Factors

Inch/Pound to International System of Units

Multiply	By	To obtain
	Length	
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
	Area	
square foot (ft^2)	0.09290	square meter (m^2)
square mile (mi^2)	2.590	square kilometer (km^2)
	Volume	
gallon (gal)	3.785	liter (L)
gallon (gal)	0.003785	cubic meter (m^3)
cubic foot (ft^3)	0.02832	cubic meter (m^3)
	Flow rate	
foot per second (ft/s)	0.3048	meter per second (m/s)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)

International System of Units to Inch/Pound

Multiply	By	To obtain
	Length	
millimeter (mm)	0.03937	inch (in.)
meter (m)	3.281	foot (ft)
kilometer (km)	0.6214	mile (mi)
	Area	
square meter (m^2)	10.76	square foot (ft^2)
square kilometer (km^2)	0.3861	square mile (mi^2)
	Volume	
cubic meter (m^3)	264.2	gallon (gal)
liter (L)	61.02	cubic inch (in^3)
cubic meter (m^3)	35.31	cubic foot (ft^3)
	Flow rate	
cubic meter per second (m^3/s)	35.31	cubic foot per second (ft^3/s)
cubic meter per second per meter [(m^3/s)/m]	35.31	cubic foot per second per meter [(ft^3/s)/m]
meter per second (m/s)	3.281	foot per second (ft/s)
millimeter per minute (mm/min)	0.03937	inch per minute (in/min)
	Mass	
kilogram (kg)	2.205	pound avoirdupois (lb)
	Density	
kilogram per cubic meter (kg/m^3)	0.06242	pound per cubic foot (lb/ft^3)
meter squared per year (m^2/yr)	10.76	foot squared per yr (ft^2/yr)
	Hydraulic gradient	
meter per kilometer (m/km)	5.27983	foot per mile (ft/mi)

Datums

Vertical coordinate information is referenced to the North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to the North American Datum of 1983 (NAD 83).

Elevation, as used in this report, refers to distance above the vertical datum.

Supplemental Information

Concentrations of chemical constituents in water are given in either milligrams per liter (mg/L) or micrograms per liter (µg/L)

Abbreviations

DIN	dissolved inorganic nitrogen
DOC	dissolved organic carbon
DON	dissolved organic nitrogen
D/T	displacement ratio
FBOM	fine benthic organic matter
HL	hydraulic load
N	nitrogen
N_2	nitrogen gas
NH_4^+	ammonium
NHD	National Hydrography Dataset
NO_2^-	nitrite
NO_3^-	nitrate
NWIS	National Water Information System
P	phosphorus
PAH	polycyclic aromatic hydrocarbon
PNW	Pacific Northwest
PO_4^{-3}	orthophosphate
Q/w	specific discharge
Sw	uptake length
TN	total nitrogen
TP	total phosphorus
U	uptake rate
USGS	U.S. Geological Survey
v_f	uptake velocity
w:d	width to depth ratio

Nutrient Attenuation in Rivers and Streams, Puget Sound Basin, Washington

By Rich W. Sheibley, Christopher P. Konrad, and Robert W. Black

Abstract

Nutrients such as nitrogen and phosphorus are important for aquatic ecosystem health. Excessive amounts of nutrients, however, can make aquatic ecosystems harmful for biota because enhanced growth and decay cycles of aquatic algae can reduce dissolved oxygen in the water. In Puget Sound marine waters, low dissolved oxygen concentrations are observed in a number of marine nearshore areas, and nutrients have been identified as a major stressor to the local ecosystem. Delivery of nutrients from major rivers in the Puget Sound Basin to the marine environment can be large. Therefore, it is important to identify factors related to how nutrients are retained (attenuated) within streams and rivers in the Puget Sound Basin. Physical, chemical, and biological factors related to nutrient attenuation were identified through a review of related scientific literature.

Numerous empirical modeling approaches for estimating nutrient attenuation in streams and rivers also were compiled, and a subset of these models was applied to the Puget Sound Basin. In particular, models based on the physical characteristics of a river reach (RivR-N model) and on the physical and biological features of a river reach (v_f model) were used and compared for the 17 major rivers draining to the Puget Sound. Data on the relative amount of instream attenuation (the fraction of nutrient input removed per kilometer of stream reach) showed some common and general themes. Firstly, headwater reaches throughout the Puget Sound Basin tend to be better than the main stems of the major rivers at attenuating nitrate (NO_3^-) and orthophosphorus (ortho-P). Secondly, rivers are more efficient at attenuating NO_3^- than ortho-P, probably because of the close relation between phosphorus and suspended sediment, which was not captured fully in the models. Thirdly, when comparing the RivR-N and v_f models for NO_3^-, physical characteristics of the channel may be more effective predictors of relative NO_3^- attenuation for main stem reaches, whereas biological factors may be more effective predictors in headwater reaches. These results are explained in terms of four primary factors of attenuation: sinuosity, channel slope, specific discharge, and uptake velocity (v_f) of the reach.

A simple scoring procedure based on these four factors showed that reaches where attenuation scores were high had higher relative attenuation of nutrients from the RivR-N and v_f models. This attenuation “scorecard” can be used to quickly assess the potential for a given reach to attenuate nutrients. Seasonal relative attenuation at three case studies was greater in summer months (July through September) and much lower and almost constant from January through June. An analysis of relative attenuation across a range of nutrient concentrations showed that, at some point, relative instream attenuation is minimized. For NO_3^-, relative attenuation reached a minimum value greater than 3 milligrams of nitrogen per liter (mg N/L) during low flow and 1 mg N/L during high flow. For orthophosphate, minimum relative attenuation was observed at about 0.1 milligram of phosphorus per liter (mg P/L) for both low- and high-flow conditions. Generally, the temporal dynamics of nutrient attenuation are dependent on the travel time through a given reach, the proportion of flow in contact with the sediment, and the amount of biological activity. Improved understanding of nutrient attenuation in Puget Sound Basin will benefit from the compilation of more detailed data for specific discharge, channel slope, and channel sinuosity in Puget Sound streams and rivers. Additionally, field studies examining upstream-downstream changes in nutrient load and field-based measurements of v_f are needed.

From a management perspective, preservation and improvement of instream nutrient attenuation should focus on increasing the travel time through a reach and contact time of water sediment (reactive) surfaces and lowering nutrient concentrations (and loads) to avoid saturation of instream attenuation and increase attenuation efficiency. These goals can be reached by maintaining and restoring channel-flood plain connectivity, maintaining and restoring healthy riparian zones along streams, managing point and nonpoint nutrient loads to streams and rivers, and restoring channel features that promote attenuation such as the addition of woody debris and maintaining pool-riffle morphologies. Many of these management approaches are already being undertaken during projects aimed to restore quality salmon habitat. Therefore, there is a dual benefit to these projects that also may lead to enhanced potential for nitrogen and phosphorus attenuation.

Introduction

Nutrients such as nitrogen (N) and phosphorus (P) are key components for all living things because they are the building blocks of proteins, enzymes, amino acids, and other organic compounds. In aquatic systems, certain amounts of nutrients are needed to sustain healthy ecosystems; however, excessive amounts of nutrients in these systems can cause the degradation of streams, rivers, lakes, and estuaries (Dubrovsky and others, 2010; Roberts and others, 2014). Nutrients, particularly N, have been identified as major stressors to the Puget Sound ecosystem. In marine waters, excess N can cause algal blooms that eventually die off and, through the decomposition process, drive dissolved oxygen to levels too low to support aquatic life. In fresh waters, N and (or) P can increase plant and algal biomass, which can also lead to low levels of dissolved oxygen. These coupled processes (increased biomass production and subsequent decay and oxygen depletion), called eutrophication, are a widespread environmental problem that is observed in many parts of the world. Low dissolved-oxygen concentrations have been observed in numerous locations in Puget Sound (Roberts and others, 2014), and riverine inputs of N to Puget Sound marine waters can be significant (Mohamedali and others, 2011), especially in south and central Puget Sound. Therefore, it is important to understand factors related to stream and river nutrient attenuation so that managers can preserve features of streams and rivers that will help reduce nutrient loads to marine waters of Puget Sound. Nutrient attenuation, also referred to as nutrient retention, is a generic term that refers to the loss of nutrient load from the stream or river as it moves downstream. This loss can be temporary, as when nutrients cycling within a stream reach and are taken up and released multiple times as they move downstream, or the loss of nutrient can be permanent. Throughout this report, the terms attenuation and retention are used interchangeably.

It has been long recognized that rivers and streams are not just pipes that move solutes downstream to recipient water bodies (Bencala, 1983). Instead, rivers and streams transform and attenuate nutrient loads through a combination of physical and biological processes as they move downstream. Point source management of nutrients can be expensive, and nonpoint source management can be challenging given the distributed nature of the sources. Understanding what factors contribute to high or low attenuation and where attenuation is likely to occur can provide an additional nonpoint source management tool. Nonpoint source management could broaden from reducing sources to enhancing and preserving factors associated with attenuation in the watershed, flood plain, and river network. This is important because reducing nonpoint sources may not result in lower levels of nutrients in river networks if attenuation characteristics are lost.

Purpose and Scope

The purpose of this report is to identify what factors are responsible for attenuating nutrients in freshwater systems and where these processes are expected to occur in order to better understand and manage for the delivery of nutrients from Puget Sound watersheds to nearshore marine ecosystems. Recent analyses of ambient monitoring data for rivers draining into Puget Sound (Mohamedali and others, 2011) suggest that summer N loads generated in Puget Sound watersheds are significantly attenuated before reaching Puget Sound. If so, then watershed and instream processes have a major mitigating effect that has not been quantified at the Puget Sound scale to date. Additionally, most pollution-reduction approaches do not account for instream attenuation when establishing load reduction targets. A review of the available scientific literature identified important factors related to nutrient attenuation in rivers and streams, and modeling approaches that can be used to estimate attenuation in river reaches and watershed river networks. The literature review presented in this report is meant to be an overview of the state of the science relevant to Puget Sound systems; however, it is not intended to serve as a detailed review for each factor related to nutrient attenuation. Nutrient attenuation in rivers and streams is complex and is controlled by numerous factors, some of which are fields of science on their own. From the literature review, attenuation models were used to identify areas of Puget Sound where high and low nutrient attenuation would be expected across different spatiotemporal scales. The focus of this report is on instream nutrient attenuation and not watershed (terrestrial) attenuation of nutrients, though it is well known that both terrestrial and instream attenuation are important for reducing total nutrient inputs to a watershed. This report was done in cooperation with the Washington State Department of Ecology (Ecology) and supports the Toxics and Nutrients Reduction Strategy developed by Ecology (Washington State Department of Ecology, 2013a) to implement a National Estuary Program grant for Puget Sound toxics and nutrients.

Study Area

The Puget Sound Basin encompasses the 13,700-mi^2 area that drains to Puget Sound and adjacent marine waters. Included in the basin are all or part of 13 western Washington counties, as well as the headwaters of the Skagit River and part of the Nooksack River in British Columbia, Canada. Streams and rivers drain three physiographic provinces—the Olympic Mountains in the west, the Cascade Range in the east, and the Puget Lowlands in the center of the basin (fig. 1).

Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 1. Study location and associated hydrologic network, Puget Sound Basin, Washington.

Nearly 4 million people, about 70 percent of Washington State's population, live in the Puget Sound Basin. Urban growth is rapid; by 2020, the population is expected to increase by 1.1 million people, with most growth in urban and suburban areas. Urban and agricultural lands, which cover about 9 and 6 percent of the basin, respectively, are concentrated in the lowlands. Forestry dominates land use in the basin and is concentrated in the foothills and mountains (fig. 2).

Forestry, agriculture, and urban development have affected the quality of freshwater in the Puget Sound Basin. The quality of groundwater in the upper watersheds probably differs little from natural conditions (Ebbert and others, 2000) but, because of development, many streams in the Puget Lowlands have undergone changes in structure and function, with a trend toward simplification of stream channels (channelization, disconnection from flood plains) and loss of aquatic plants and organism habitat (Black and Silkey, 1998; Collins and Montgomery, 2002; Collins and others, 2002; Collins and others, 2012).

Heavy industry generally is located on the shores of the urban bays and along the lower reaches of their influent tributaries, such as Commencement Bay and the Puyallup River in Tacoma and Elliott Bay and the Duwamish Waterway in Seattle. High density commercial and residential development occurs primarily within and adjacent to the major cities. This trend has resulted in increasing urban sprawl in the central Puget Sound Basin, and urban land-use activities have had a significant effect on the quality of streams in the Puget Sound Basin (May and others, 1997; Staubitz and others, 1997; Morley and Karr, 2002; McBride and Booth, 2005; Herrera Environmental Consultants, Inc., 2011). Water-quality concerns related to urbanization include providing adequate sewage treatment and disposal, transporting contaminants to streams by storm runoff, and preserving stream corridors. It is not uncommon, particularly during high flow events, for measurable concentrations of herbicides, pesticides, fungicides, industrial chemicals, metals, polycyclic aromatic hydrocarbon (PAHs), and (or) elevated nutrients to be in the highly urbanized Puget Sound lowland rivers and streams (Ebbert and others, 2000; Herrera Environmental Consultants, Inc., 2011; Conn and Black, 2014).

More than half of the agricultural acreage in the basin is located in Whatcom, Skagit, and Snohomish Counties. Agricultural land use consists of about 60 percent cropland and 40 percent pasture. Livestock produce a large amount of manure that is applied as fertilizer to cropland, sometimes in excess amounts, resulting in the runoff of N and P to surface water and the leaching of nitrate to groundwater. Runoff from agricultural areas also carries sediment, pesticides, and bacteria to streams (Staubitz and others, 1997; Ebbert and others, 2000).

Sources of contaminants to Puget Sound lowland streams and the lower reaches of large rivers are largely nonpoint because most major point sources discharge directly to Puget Sound (Ebbert and others, 2000). The concentration of anthropogenic chemicals in the water of the lower reaches of large Puget Sound rivers are often lower than that in small streams originating in the Puget Lowlands because much of the flow is derived from the forested and alpine headwaters. However, during high flows events, particularly during late summer and early autumn, concentrations of nutrients and other anthropogenically generated chemicals can increase (Ebbert and others, 2000; Conn and Black, 2014).

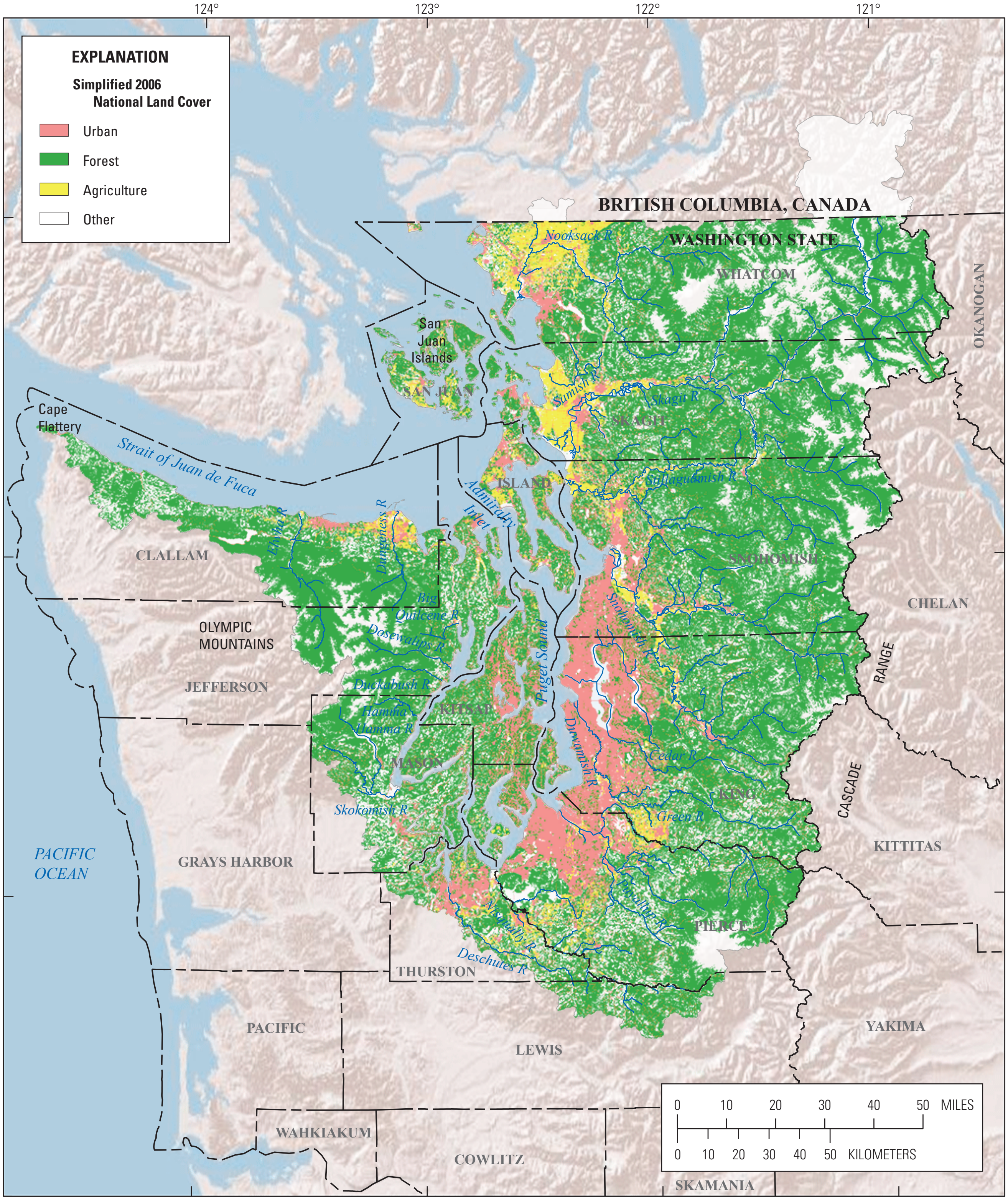
Nutrient Cycling in Streams and Rivers

During transport downstream, N and P undergo complex cycles, which can be thought of as nutrient spirals that are constantly transforming nutrients as water moves from the headwaters to receiving water bodies. To understand the factors that lead to nutrient attenuation in streams and rivers, it is important to first understand the sources and cycles of N and P that take place in fluvial systems.

Sources and Cycles of Nitrogen

In freshwater systems, N occurs in many forms, including dissolved and particulate fractions. Dissolved inorganic nitrogen (DIN) is made up of nitrate (NO_3^-), ammonium (NH_4^+), and nitrite (NO_2^-); dissolved organic nitrogen (DON) is made up of complex organic compounds such as amino acids; and particulate N occurs in organic form as microbes and detritus (leaf litter and other decaying organic matter). It is the dissolved inorganic forms (NO_3^- and NH_4^+) that contribute the most to water quality problems related to eutrophication and hypoxia (Rabalais and others, 1996) because aquatic plants, algae, fungi, and bacteria assimilate, or take up, N primarily as NH_4^+ and NO_3^- (Duff and Triska, 2000).

Sources of N to streams include atmospheric deposition, N fixation, and terrestrial inputs carried from runoff and groundwater flow to the stream, and are comprised of natural and anthropogenic sources. As a result, the amount and form of N source to a stream is highly dependent on local land use (Allan and Castillo, 2007; Dubrovsky and others, 2010) as the contribution of natural and anthropogenic sources also varies across land use. Atmospheric deposition consists of precipitation (wet deposition) and dry fallout (dry deposition) and is mainly in the form of NH_4^+ and NO_3^-. Nitrogen fixation is the conversion of relatively inert nitrogen gas (N_2) into more biologically available forms and is mainly a natural biological (bacterial fixation through cyanobacteria) and chemical (the high energy from lightning strikes can result in fixation) process. However, with the increase in synthetic fertilizer production, the burning of fossil fuels, and the cultivation of N-fixing crops such as soybeans, anthropogenic N-fixation now greatly exceeds natural fixation (Vitousek and others, 1997).



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 2. Simplified land cover, Puget Sound Basin, Washington, 2006. From Fry and others (2011).

Terrestrial inputs can be in dissolved forms from runoff and groundwater pathways and in particulate forms such as litter fall (Roberts and Bilby, 2009). Direct inputs to the stream of N from precipitation and through fall (water that drips from vegetation) are less important (Duff and Triska, 2000) in most cases but can be significant at the beginning of a rain event (Allan and Castillo, 2007). Atmospheric deposition to the watershed, however, can be a large source of N that can make its way into streams through groundwater flow (Shibata and others, 2001). Generally, because the main reservoir of N in the environment is the atmosphere, rock weathering as a source of N is assumed negligible, although in some cases it can be important (Holloway and others, 1998). The main anthropogenic inputs of N to streams include agricultural and urban fertilizers in runoff, atmospheric deposition from industrial fertilizer production and fossil fuel burning, and wastewater from direct (treatment plants) and indirect (septic systems) sources. In Puget Sound, sources of N to streams are primarily from point sources, livestock manure, runoff from forest and developed land uses, and N-fixing alder trees (Bechtold and others, 2003; Brett and others, 2005a; Herrera Environmental Consultants, Inc., 2011; Steinberg and others, 2011; Wise and Johnson, 2011).

The N cycle in streams and rivers is complex, with most reactions being microbially mediated (fig. 3). Generally, reactions are either used to obtain N for structural synthesis (assimilatory uptake) or for energy in microbial reactions (dissimilatory uptake). Atmospheric N_2, the most abundant form of N globally, is soluble in water and therefore can exchange to and from the water column in streams. In streams and rivers, dissolved N_2 can be converted into oxidized forms (NH_4^+ and NO_3^-) by N-fixing bacteria (cyanobacteria), through N fixation (fig. 3), and incorporated into cells. The oxidized N not used by the N-fixers is released

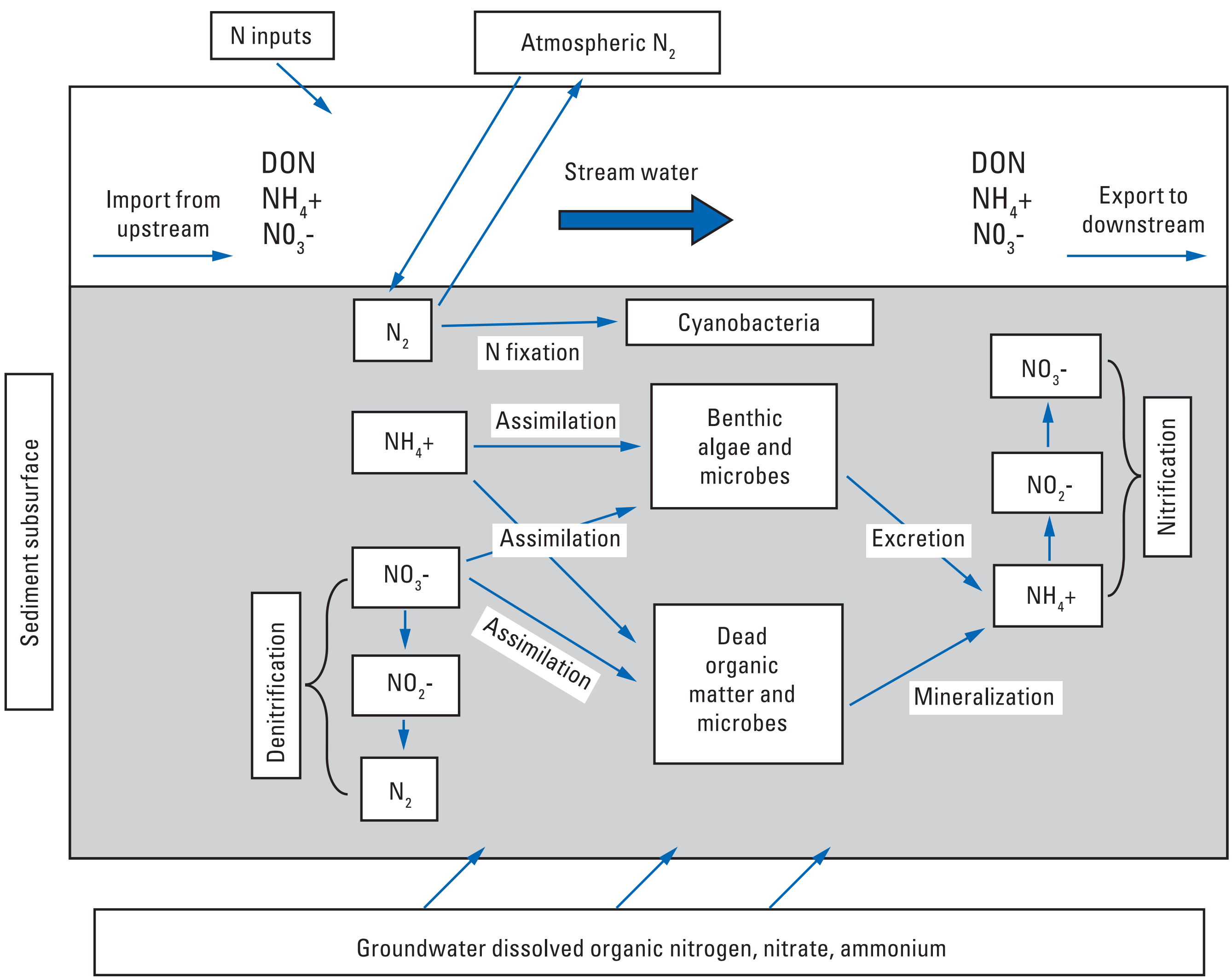


Figure 3. Nitrogen (N) cycle in streams and rivers (adapted from Allan and Castillo, 2007). DON, dissolved organic nitrogen; NO_3^-, nitrate; NH_4^+, ammonium; NO_2^-, nitrite; N_2, nitrogen gas.

(excreted). Although N-fixation is important in some streams (Grimm and Petrone, 1997; Kunza and Hall, 2014), most N inputs into a stream come from riparian vegetation (litter fall), overland flow, and groundwater flow.

Both NH_4^+ and NO_3^- are easily assimilated by benthic algae, aquatic plants, and microbes associated with decaying organic matter. Generally, NH_4^+ is more energetically favorable and is taken up more readily than NO_3^-. Breakdown of organic matter represents the opposite process, where organic N is mineralized to NH_4^+. Besides direct assimilation of N by plants and microbes, there are two important energy-yielding microbial reactions that are common in streams and rivers, nitrification and denitrification. Nitrification is a microbial process that converts NH_4^+ to NO_3^- and is favored under more aerobic conditions (fig. 3). Denitrification converts NO_3^- to N_2 and denitrifying bacteria can thrive under both low and high oxygen conditions; however, they use NO_3^- only for energy production when oxygen is low (fig. 3). Denitrification is one of the most important N cycling processes because the end product, N_2, is inert to most living things and can be released into the atmosphere, representing a true loss of N from stream and river ecosystems. Although not as common as nitrification and denitrification, the process of dissimilatory NO_3^- reduction to ammonium (DNRA) can also take place. Under certain conditions, NH_4^+ can actively exchange (sorb and desorb) with cation sites within the sediment matrix representing a temporary loss or storage of N. As shown in figure 3, most N cycling takes place within streambed sediments, and uptake directly from the water column is believed to be a small amount of total uptake; however, some studies have shown that, in large rivers, uptake from the water column can be important (Tank and others, 2008).

Sources and Cycles of Phosphorus

Phosphorus can be present in stream water as orthophosphate (PO_4^{-3}) dissolved in water and attached to inorganic particles in suspension, as dissolved organic molecules, and as particulate organic forms from bacteria and detritus. Generally, P exists in greater proportions in particulate forms compared to dissolved forms; therefore, landscape features such as soil type, geology, terrestrial vegetation, topography, land use, along with human sources control the amount and form of P in streams and rivers (Hendricks and White, 2000). Most P in soils and surrounding geology is insoluble; however, as these materials erode and become weathered, there is a potential for this P to enter streams and become more readily available. This is because PO_4^{-3} easily adsorbs to charged particles, especially clays, which are transported to streams in runoff. The decomposition of litter from terrestrial systems further adds to the dissolved and particulate forms of inorganic P from runoff and groundwater discharge (Roberts and Bilby, 2009).

Atmospheric deposition of P is lower than deposition of N (Allan and Castillo, 2007); however, similar to N, local land use has a big effect on the P loading to streams and rivers (Brett and others, 2005a, 2005b; Herrera Environmental Consultants, Inc., 2007, 2011). Anthropogenic sources of P to streams and rivers are from point source wastewater discharges, and non-point sources from fertilizer and manure applications, and failing septic systems (Brett and others, 2005a; Withers and Jarvie, 2008). These nonpoint sources can enter streams and rivers by runoff and groundwater flowpaths. Wastewater sources can have a big influence on river water quality and primary productivity because a large fraction of these inputs is as PO_4^{-3}, which is taken up easily and efficiently by stream biota. Similarly, runoff can include a larger fraction of dissolved P, but also carries particulate forms as soil and sediment are mobilized during runoff events. In Puget Sound, direct wastewater discharges of P are not important (Ebbert and others, 2000; Brett and others 2005a); however, non-point source inputs of P to streams from urban and agricultural land uses (Brett and others, 2005a, 2005b; Herrera Environmental Consultants, Inc., 2007, 2011) and geologic material (Wise and Johnson, 2011) are significant.

Phosphorus cycling in streams is complex (fig. 4) and influenced by physical, chemical, and biological processes. The main biological processes include the assimilation of dissolved inorganic P into cells of phytoplankton, macrophytes, and benthic algae; transfer of this organic P through the food chain to invertebrates and fish, and then released again by excretion and remineralization during decomposition (fig. 4). Algae and microbes in biofilms obtain most of their P from the water column, whereas rooted plants will acquire their P from sediments and sediment pore waters. Physical-chemical processes such as sorption and desorption and complexation also influence the amount of available P in streams and rivers. Sorption and desorption are dependent on the capacity of sediments to retain P and can influence local diffusional gradients at the sediment-water interface. Complexation, or bonding, of P to metal oxides and hydroxides under aerobic conditions can release P when conditions become more anaerobic. This process can result in the temporary storage of P at sediment surfaces, which becomes available later when sediments are buried and soluble P is released into pore waters. The physical attributes of a channel, such as slope and presence of pools and riffles create areas for sediment transport and deposition, which temporarily store and release particulate P as it moves downstream (Reddy and others, 1999).

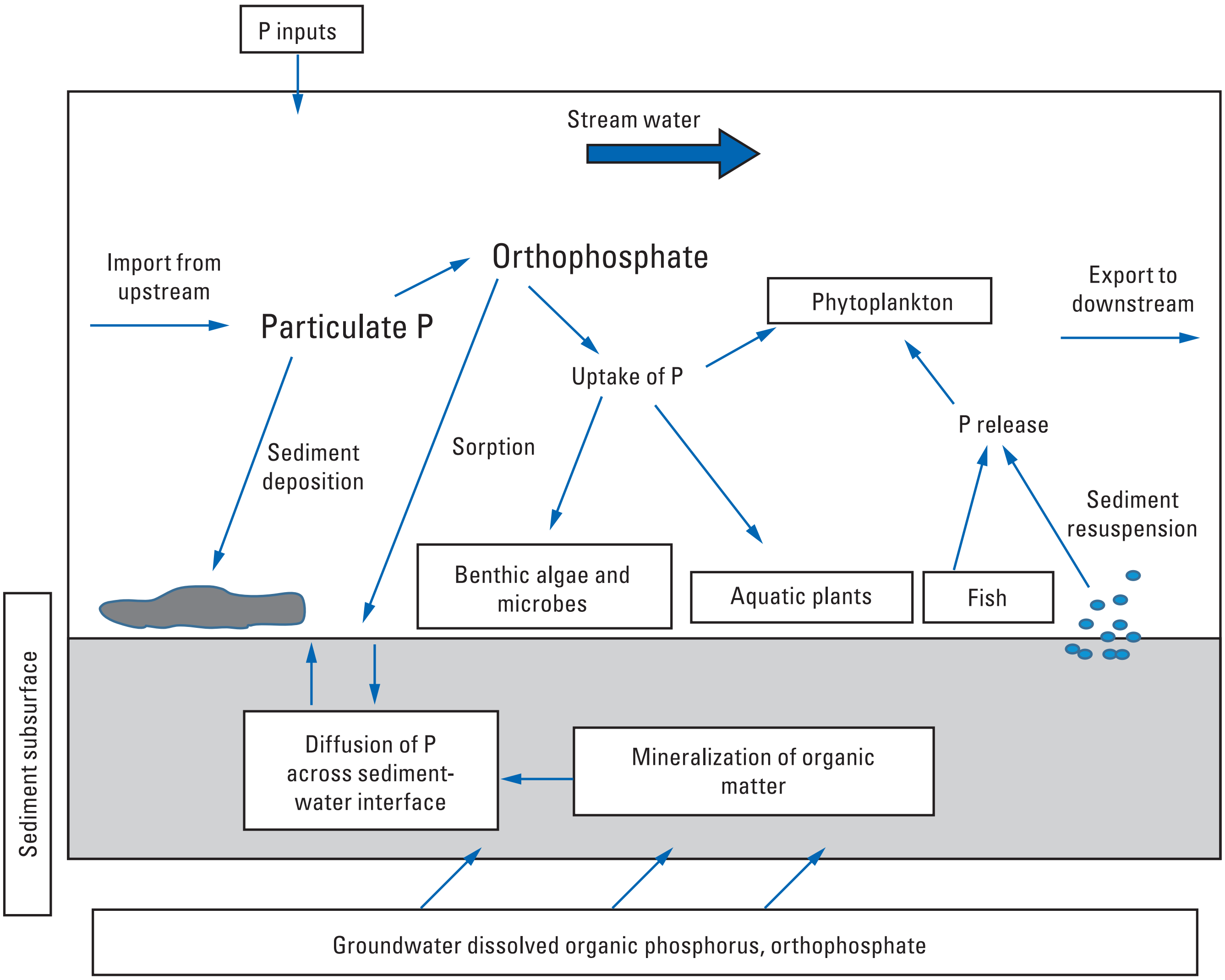


Figure 4. Example of the phosphorus (P) cycle in streams (adapted from Allan and Castillo, 2007).

Factors Related to Nutrient Attenuation in Streams and Rivers

The N and P cycles in streams are dictated by a complex suite of physical, chemical, and biological processes that make these cycles dynamic in space and time. Flow in streams and rivers is dominated by movement in the downstream direction; therefore, advection of dissolved and particulate forms of N and P results in a constant interaction of nutrients with surface and subsurface features of these environments. The factors related to nutrient attenuation in streams and rivers are numerous and vary in scale from the watershed, to the reach level, to individual channel features. Furthermore, factors of nutrient attenuation are intricately connected where biological factors can be influenced by physical factors, biological factors can be influenced by chemical factors, and physical factors can be influenced by biological factors. This report summarizes the important physical, chemical, and biologic factors related to nutrient attenuation. Additionally, although the focus of this report is on the important instream factors related to nutrient attenuation, important watershed factors related to nutrient delivery to streams and rivers are briefly summarized.

Physical Factors

Physical factors of nutrient attenuation are factors of a stream or river reach related to the hydrologic and geomorphic characteristics of the channel. The main overarching theme in the scientific literature is that increasing the time nutrient molecules spend within a given river reach enhances the ability of that reach to retain or attenuate these nutrients. Streamflow, water velocity, and channel geometry (width and depth) are interrelated and contribute to travel time of a stream or river reach. In the simplest form, the slower the surface water velocity, the longer the travel time. Because streamflow is defined as the velocity of the water times the cross-sectional area of the channel (width times the depth), changes in any of these parameters will result in changes to the average reach travel time. Several studies have shown that small streams (1st and 2nd order) are more efficient at nutrient retention than large streams (4th order and greater), particularly when looking at the watershed scale (Alexander and others, 2000; Bernot and Dodds, 2005; Ensign and Doyle, 2006; Wollheim and others, 2006; Mulholland and others, 2008; Roberts and others, 2008). Generally, there are more stream miles of small streams in river networks than in larger order streams and these small streams have lower flows and greater surface areas per unit flow than large streams that have high velocities and are more channelized. The width to depth ratio of a reach also is important because it is related to how much of the flow is in contact with the stream bottom, which is an active area for nutrient processing.

Channel-flood plain connectivity is an important physical feature of stream and river reaches related to attenuation in a given watershed. The more connected a channel is with its flood plain, the more opportunity there is for high streamflows to access the flood plain with its shallow topography resulting in slower velocities and increased travel times. The interaction of a stream channel with its flood plain has been shown to retain water, sediment, and nutrients (Jansson and others, 1994). The main pathway for N loss is through denitrification (Mitsch and others, 2000; Saunders and Kalff, 2001; Sheibley and others, 2006), whereas the main pathway for P loss is sedimentation (Behrendt and Opitz, 1999; Olde Venterink and others, 2003). The nutrient retention capacity of a flood plain depends on many factors such as soil properties, availability of labile organic carbon (Davidsson and Ståhl, 2000), water temperature (Pinay and others, 2007), soil moisture (Olde Venterink and others, 2003), flow velocities, and water retention times (Tockner and others, 1999). All these properties are tied to the frequency and duration (and extent) of flooding within a given flood plain unit when relating to how much nutrient attenuation can take place (Natho and Venohr, 2014). Flood plains can be important for nutrient retention over an annual time scale because increased flows during storms can carry a significant amount of the annual load of nutrients in many watersheds (Ahearn and others, 2004; Macrae and others, 2007). Channel confinement, the ratio of the flood plain width to the channel width (Knighton, 1998), is commonly used to evaluate channel-flood plain connectivity. When the channel confinement is greater than three the channel is considered unconfined and there is a potential for substantial exchange of water between channel and flood plain, which can lead to increased retention of organic matter (Bellmore and Baxter, 2013).

In addition to channel-flood plain exchange of water, streams and rivers continuously exchange water between the main channel and subsurface sediment, surrounding riparian areas, and underlying groundwater (Boano and others, 2014). The exchange of water in the vertical and lateral directions with slower moving water beneath and along the stream slows the bulk flow of dissolved and suspended materials (termed transient storage) that can increase the contact time with reactive surfaces (Jones and Mulholland, 2000). Transient storage can occur as surface storage in the main channel of rivers and streams as a result of pools, side channels, back waters, and eddies, or as a result of hyporheic exchange. Hyporheic exchange, also called hyporheic transient storage, is the transport of surface water through subsurface flowpaths below and lateral to the main channel that returns to surface water; this exchange may or may not also interact with the shallow groundwater system (Bencala and Walters, 1983; Triska and others, 1989; Harvey and others, 1996). Therefore, the total amount of transient storage in a given reach is made up of both surface and hyporheic components. Transient storage zones have been shown to greatly affect N and P cycling because they extend residence times within the reach and increase exposure of surface water to areas of biologically active sediment communities (bacteria, plants, algae) (Mulholland and others, 1997; Duff and Triska, 2000; Hendricks and White, 2000; Briggs and others, 2009). Although the correlation between increases in transient storage and increases in nutrient retention is not universal (Ensign and Doyle, 2006), generally, features of a stream or river that enhance transient storage, in particular hyporheic storage, will likely result in increased nutrient attenuation.

Over the last few decades, there has been a great deal of research associated with understanding the physical and geomorphic controls of hyporheic exchange (see Boano and others [2014] for a recent review). Generally, there is a greater potential for hyporheic exchange in a stream or river that contains heterogeneous habitat elements at both large scales (for example, pool-riffle sequences and secondary channels) and small scales (heterogeneous substrate, debris dams, and in-channel vegetation). Hyporheic exchange in streams has been shown to increase in the presence of large woody debris (Valett and others, 2002), in-stream vegetation (Harvey and others, 2003), channel meanders (Boano and others, 2006), and pool-riffle sequences (Gooseff and others, 2006; Tonina and Buffington, 2007). In addition to these large channel features, sandbars and sand dunes (Thibodeaux and Boyle, 1987; Holmes and others 1994; Elliot and Brooks, 1997; Tonina and Buffington, 2011; Hester and others, 2013; Stonedahl and others, 2013), and coarse substrate such as cobbles (Vaux, 1968) have been shown to promote hyporheic exchange. A growing body of literature shows that restoration of in-stream features such as constructed riffles, steps, debris dams, and weirs can promote N processing (Shibata and others, 2004; Ensign and Doyle, 2005; Kasahara and Hill, 2006a, 2006b). Physical characteristics of stream and river reaches such as channel slope and sinuosity, channel roughness, and substrate size have been determined to be important factors that contribute to hyporheic exchange and in turn, nutrient retention (Gooseff and others, 2007; Baker and others, 2012).

Urbanization in the Puget Lowlands has resulted in changes to stream channels and their associated hydrologic characteristics throughout the region. The increase in amount of impervious surfaces can lead to quick movement of stormwater to streams and rivers, increase the flashiness of a stream, and increase the peak discharge of storms (Booth and others, 2002; Konrad and others, 2005). Flood control practices in urban streams can result in the removal of important channel features such as woody debris. Channel straightening (Booth and others, 2002) can result in a loss of habitat complexity. As a result, urbanization results in higher stream water velocities, less dispersion, and shorter travel times (Roberts and Bilby, 2009) compared to forested streams as well as the loss of important channel features that promote hyporheic exchange. Little information exists for characterizing important physical factors for nutrient attenuation directly in Puget Lowland streams. However, several habitat monitoring programs likely gather relevant data, which have not been summarized or widely distributed.

Chemical Factors

Chemical factors related to nutrient attenuation include the amount of nutrients present, the amount of oxygen present, and for some N cycle processes, the amount of organic carbon present. Multiple field and laboratory based studies have determined that the rate of nutrient uptake is a function of the nutrient concentration (Dodds and others, 2002; Inwood and others, 2005; Bernot and others, 2006; Mulholland and others, 2008) and the more nutrients that are present, the greater the potential for attenuation. However, this relation is not linear, and a leveling off, or saturation, of retention is typically found at high nutrient concentrations (Dodds and others, 2002). Although the rates of nutrient attenuation may increase in high nutrient loaded systems, the proportion of the stream or river load retained decreases as concentrations increase, reducing the efficiency of overall attenuation (Inwood and others, 2005; Mulholland and others, 2008; Alexander and others, 2009). Many of the processes in the N and P cycles are controlled by the amount of oxygen in surface water and hyporheic waters (figs. 3 and 4). For example, denitrification, which represents a true loss of inorganic N from the system, needs anaerobic conditions in the sediment to take place. For denitrification, organic carbon also is required as an energy source for denitrifying bacteria. Therefore, low oxygen conditions in the sediment, often related to anaerobic microsites (Sheibley and others, 2003a), the amount of dissolved organic carbon (DOC), and presence of fine benthic organic matter (FBOM) are important factors related to N retention (Mulholland and others, 2000; Meyer and others, 2005; Inwood and others, 2007; Harvey and others, 2013). Thus, as the overall percentage of fines in a given stream reach increases, N attenuation also is expected to increase. Likewise, the sorption/desorption of P is highly dependent on the redox conditions present in the sediment.

Biological Factors

Biological processes of N attenuation in streams and rivers includes both microbial and algal/plant uptake processes. Biological uptake for P is greatest through plant/algal uptake processes; therefore, factors that result in increases in plant and algal biomass will be important for nutrient attenuation. Firstly, because algae and plants are primary producers, they need sunlight and as the amount of sunlight increases, the gross primary production (GPP) in streams increases (Mulholland and others, 2001). Secondly, studies have shown that nutrient attenuation is greater during daylight (Fellows and others, 2006; Mulholland and others, 2006) and N and P retention is correlated with gross primary production (GPP) and community respiration (CR) of streams (Mulholland and others, 2001; Hall and Tank, 2003; Meyer and others, 2005; Roberts and Mulholland, 2007).

Plant and algal communities also have an indirect effect on nutrient attenuation. The presence of large algal mats and aquatic plants changes the hydraulic characteristics of the stream by increasing dispersion coefficients and transient storage, and enhancing nutrient uptake (Mulholland and others, 1994; Sand-Jensen and Mebus, 1996; White and Hendricks, 2000). These biological features increase travel times through a reach and can trap sediment over time, which provides a source of organic matter for microbial processing of N and alters stream bed topography to promote increased hyporheic exchange (White and Hendricks, 2000; Duff and others, 2002).

Biological processes are also influenced by temperature, which has been shown to influence stream nutrient concentrations of N and P (Hendricks and White, 1995; Sheibley and others, 2003a; Triska and others, 2006). Warm temperatures are related to the amount of sunlight and increases in microbial uptake kinetics, thus higher temperatures tend to correlate with high rates of N and P retention across many studies (Martin and others, 2001; Sheibley and others, 2003b; Simon and others, 2005; Hanafi and others, 2006; Mulholland and others 2006; Triska and others, 2007).

Watershed Related Factors

Although the focus of this report is on instream factors related to nutrient attenuation, several factors of a stream's watershed control the amount of nutrients delivered to streams and rivers and subsequently the amount of instream nutrient attenuation. For example, watershed size correlates with the amount of N delivered to rivers and streams (Saunders and Kalff, 2001) because a large watershed receives greater amounts of N, which can potentially reach streams and rivers. Other watershed factors include human population, development (agriculture and urbanization), and the amount of impervious surfaces in a watershed, all of which are related to watershed land use. Streams and rivers in developed areas of Puget Sound are enriched with nutrients relative to those in undeveloped areas (Ebbert and others, 2000; Herrera Environmental Consultants, Inc., 2007, 2011). Recent data collected throughout the nation have shown that the highest average concentrations of total N are in small streams draining agricultural areas (Dubrovsky and others, 2010). Similar trends were observed in the Puget Sound Basin where urban and agricultural land use in the basin is positively correlated to instream dissolved inorganic nitrogen (DIN) concentrations (Roberts and others, 2014). Fertilizers used in agricultural and urban areas, manure associated with dairy farms, and atmospheric deposition are sources of N in Puget Sound Basin rivers (Embrey and Inkpen, 1998). Average concentrations of total P that exceed the goal of 0.1 mg/L to prevent excessive plant growth (U.S. Environmental Protection Agency, 1986) have been detected in streams and rivers in all land-use areas of Puget Sound Basin except undeveloped land (Ebbert and others, 2000). Unlike total N, concentrations of total P in streams and rivers in the Puget Sound Basin do not correlate with watershed application rates and atmospheric deposition of P in drainage basins (Inkpen and Embrey, 1998). Phosphorus attaches to soil particles and usually remains close to application areas unless it is transported to rivers by soil erosion. Because erosion transports P to streams, yields of total P correlate with yields of suspended sediment. The presence of riparian zones along stream corridors can influence nutrient delivery to streams (Naiman and Décamps, 1997) and the importance of riparian zones for reducing terrestrial loads of nutrients as they move through surface and subsurface flowpaths to streams and rivers has been demonstrated (see Ranalli and Macalady, 2010 for a recent review).

Numerous physical, chemical, biological, and watershed factors related to nutrient attenuation in streams and rivers are shown in table 1. Any of these factors can work together in a real world system; however, little is known about the importance of a single factor, or interaction of factors, on reach-scale nutrient attenuation. Some studies have attempted to combine multiple factors and related ecosystem processes that directly and indirectly influence reach scale nutrient attenuation (Hall and others, 2009; Mulholland and others, 2009).

Methods for Evaluating Nutrient Attenuation in Rivers and Streams

There are several methods for estimating attenuation in streams and rivers at both the stream reach and watershed scales. At the stream-reach scale, field-based methods typically are used to directly or indirectly measure attenuation rates. These field-based methods were developed for and have been largely applied to small streams; however, recent studies have tried to adapt small-stream field methods to large rivers (Dodds and others, 2008; Tank and others, 2008) with some success. At the watershed-scale, empirical relations measured at the reach-scale are incorporated into river network models. Estimating stream and river attenuation in the field at the reach-scale provides local, site-specific information about nutrient attenuation that is useful on its own as well as information that can be used to develop model parameters that are optimized for a given study area. Field methods can be labor intensive, which limits the number of sites where direct estimates can be made. Despite this challenge, when using a watershed-scale model of nutrient attenuation, having local data to feed into the models is always preferred to using literature-based data. Approaches to modeling watershed-scale attenuation vary in the extent to which they incorporate empirical reach-scale measurements, the complexity in which they represent stream and watershed hydrology, and the scale at which they predict attenuation (that is, small watershed to regional/continental).

Field-Based Methods of Instream Nutrient Attenuation

Generally, there are three approaches to estimating reach-scale attenuation in the field: (1) reach scale nutrient mass balances; (2) field or laboratory experiments that estimate nutrient uptake of individual processes, which can then be scaled up to a study reach; and (3) short-term tracer and nutrient additions. Mass balance studies measure all nutrient inputs and outputs of a given stream reach to estimate the amount of attenuation by calculating the difference between inputs and outputs. Mass balance studies sometimes ignore groundwater nutrient inputs and focus only on longitudinal change in surface water nutrient load; however, even when surface water nutrient loads are the same between upstream and downstream locations, retention may be taking place if groundwater input loads are taken into account (Duff and others, 2008; Sheibley and others, 2014).

Measuring fine-scale nutrient uptake rates using field or laboratory based experiments involve the calculation of a nutrient uptake rate per unit area of streambed, which is multiplied by the total surface area of the stream reach to estimate the total amount of reach-scale attenuation. This can be done in numerous ways, but the most common methods use benthic flux chambers (Davis and Minshall, 1999; Fellows and others, 2006), the use of sediment-water flask studies in the laboratory (Duff and Triska, 1990; Garcia-Ruiz and others, 1998; Kemp and Dodds, 2002; Inwood and others, 2007), or sediment cores in the field or laboratory (Sheibley and others, 2003a; Strauss and others, 2004; Fellows and others, 2006).

Table 1. Summary of physical, chemical, biological, and watershed factors associated with nutrient attenuation in streams and rivers.

Factor	Parameter	Effect
	Physical factors	
Substrate size/type	Sand and gravel (percent)	Increase hyporheic exchange potential.
Travel time	Discharge	High flows reduce travel time through the reach.
	Stream velocity	High velocity reduces travel time through the reach.
Channel geometry	Specific discharge (Q/w)	Low values indicate a greater proportion of flow in contact with stream bottom.
	Width to depth ratio (w:d)	Larger values indicate a greater proportion of flow in contact with stream bottom.
Flood plain connectivity	Channel confinement	Increases overall downstream travel time.
		Slows down velocity of water; increases surface area available for nutrient attenuation.
	Transient storage	
Surface storage	Backwaters/eddies Side channels Pools	These features contribute to increases in travel time in the reach, but play a less important role in attenuation compared to hyporheic exchange.
Hyporheic exchange		
Riffle pool sequences	Channel slope	Changes in slope promote subsurface flow.
Sand dunes and sand bars	Channel roughness	Changes in pressure differentials promote subsurface flow.
Large woody debris Boulders		These features promote subsurface flow below and around these structures.
	Chemical factors	
Nutrient concentration	Nitrate (NO_3), phosphate (PO_4) concentration	High nutrient concentrations result in high rates of attenuation, but the efficiency of retention decreases.
	Uptake velocity (v_f)	The uptake velocity is a measure of uptake efficiency and levels off at high concentrations.
		Increase nitrogen attenuation through denitrification.
Redox condition	Dissolved-oxygen concentration (DO)	Low DO favors denitrification and nitrogen loss.
		DO concentration is related to sorption/desorption of phosphorus in sediment.
Presence of organic matter	Percent fines (fine benthic organic matter)	
	Biological factors	
Presence of algae and plants	Sunlight	Open canopies result in greater primary productivity and more nutrient uptake.
Algal and plant alteration of hydraulics	Decreased velocities	These indirect processes take place in the presence of macrophyte beds, which can lead to greater attenuation of nutrient.
	Deposition of organic matter Formation of hummocks	
Uptake rate kinetics	Temperature	Warmer temperatures tend to result in higher rates of nutrient uptake.
	Watershed related factors	
Watershed characteristics	Watershed area Population Urban and agriculture (percent) Impervious surface (percent)	All of these factors contribute to increases in nutrient loading to rivers and streams, thereby reducing attenuation efficiency in a given reach.
	Riparian buffers	In the presence of intact riparian buffers, nutrient delivery to the reach is decreased.

One of the most common approaches for estimating nutrient attenuation in the field, particularly since the 1990s, is using short-term nutrient and stream tracer additions. This procedure involves injecting a concentrated solution of nutrient and conservative tracer into the upstream end of a reach and then measuring nutrient and tracer concentrations longitudinally at multiple downstream locations along the reach when the stream reaches equilibrium. The reach is determined to be in equilibrium when the concentration of the conservative tracer has reached a plateau at the most downstream location of the reach. A conservative tracer is a chemical that will not undergo significant transformation by biological activity, whereas the nutrient concentration will change as it moves downstream through the reach (non-conservative) as a result of various uptake processes. Data from these injection experiments are used to estimate travel time and changes in flow along a reach by analyzing the behavior of the conservative tracer. The change in nutrient concentration relative to the tracer also can be used to estimate the amount of attenuation in the reach based on the nutrient spiraling concept (Webster and Patten, 1979; Newbold and others, 1981). These experiments calculate attenuation through the use of three nutrient spiraling metrics: the uptake length (S_w), uptake rate (U), and uptake velocity (v_f) (details on performing a nutrient addition experiment are presented in Stream Solute Workshop [1990]). The S_w is the mean stream distance traveled by a nutrient atom before it is removed from the water column, U is the mass of nutrient removed from the water column per unit area of stream bottom per unit time, and v_f, the uptake velocity, is analogous to a mass transfer coefficient and describes how efficient a given reach is at attenuating nutrients. The three metrics are related to one another by the following equations (Newbold and others, 1981; Stream Solute Workshop, 1990):

$$S_w = QC/(U_w) \quad (1)$$

$$U = v_f C \quad (2)$$

$$v_f = Q/(S_w w) \quad (3)$$

where

C is surface-water nutrient concentration,
w is stream width, and
Q is discharge.

These metrics are typically determined for reaches ranging from less than 100 to several thousand meters long with the assumption that the values hold for longer reaches if the experimental reach incorporates the typical characteristics (width, depth, flow, sediment type) of the longer reach. The uptake length can be determined graphically by plotting the ratio of nutrient to tracer concentration compared with downstream distance; the slope of this line is $-1/S_w$ (fig. 5).

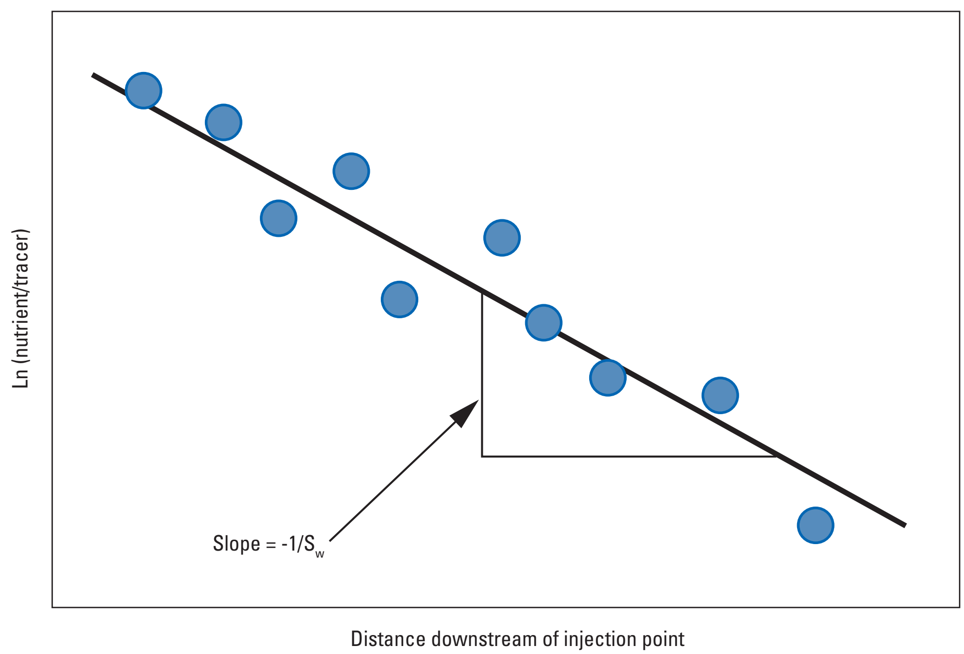


Figure 5. Example of how to determine the uptake length (S_w) for a stream reach from a short-term nutrient addition experiment. The natural log (ln) of the ratio of nutrient to tracer concentration in the stream is plotted over distance downstream of the injection point.

Once S_w is calculated, U and v_f are determined from stream width, discharge, and nutrient concentration using the equations 1–3.

The nutrient spiraling approach is described in detail because it is common to incorporate these metrics into model estimates of watershed-scale instream attenuation. The spiraling approach is limited by the assumptions that uptake follows first order kinetics and that dispersion and transient storage within the reach are negligible. Another limitation of the spiraling approach is that it does not measure true ambient or background rates of nutrient uptake because it involves an artificial addition of nutrients to the system. Because nutrient uptake is a function of nutrient concentration (see Dodds and others, 2002), nutrient additions result in an estimate of net uptake and do not reflect gross uptake of the stream or river. However, different approaches to analyzing data from a typical nutrient injection experiment (Payn and others, 2005; Earl and others, 2007; Covino and others, 2010) or the use of stable isotopes of N (Webster and others, 2003; Mulholland and others, 2008) and P (Mulholland and others, 1990) can result in better estimates of ambient rates of attenuation.

Watershed-Scale Models of Instream Nutrient Attenuation

Watershed-scale approaches to estimating nutrient attenuation use empirical models that are based on expressions derived from field-collected data directly or indirectly from a model parameterization process. These models provide expressions for reach level attenuation that are based on correlations between measured attenuation data and channel or watershed characteristics; estimates based on summarized scientific literature; or estimates derived from expressions that describe changes in nutrient concentrations in reaches as water moves downstream. Watershed-scale models are independent in space so an attenuation expression can be used across study areas but will vary in complexity based on the representation of watershed hydrology and the amount of data needed to parameterize the model. For example, models for nutrient attenuation can range from simple expressions based on only physical features of a stream channel (Seitzinger and others, 2002) to complex watershed models that simulate flow and water quality using detailed water quality sampling in the study area of interest in order to calibrate and fit model parameters used to describe attenuation (Smith and others, 1997; Arheimer and Brandt, 1998; Jung and Deng, 2011). A brief summary of empirical models of nutrient attenuation is provided in table 2. There is a wide range of complexity represented in these models; however, a common feature of all these models is that the residence time of water within a given reach is used as a key factor influencing the amount of attenuation that takes place within a reach.

Analysis of Relative Nutrient Attenuation in Puget Sound Basin

Analysis of relative nutrient retention in the Puget Sound Basin was done at multiple scales to better understand the most important factors related to attenuation in the region. The Puget Sound region lacks a comprehensive assessment of nutrient spiraling metrics (v_f, S_w, U) or other field-based estimates of nutrient attenuation (nutrient-tracer injections, mass balances, measurement of sediment uptake rates). Although SPARROW (a watershed-scale spatially explicit regression model that can estimate instream attenuation) has been used to assess the Puget Sound region as part of a larger application to the Pacific Northwest (PNW) (Wise and Johnson, 2013), the model has not been optimized for Puget Sound conditions. Therefore, as an initial analysis of nutrient attenuation patterns in the Puget Sound Basin, the potential N and P removal across Puget Sound stream reaches was estimated using two empirical attenuation models; a framework was developed to identify and score stream reaches with potentially high attenuation capacity; and temporal drivers of N and P attenuation were analyzed in specific Puget Sound stream reaches. This assessment uses only information that can be expressed as mapped characteristics to explain the relative attenuation of nutrients in Puget Sound rivers and streams based on best available information. Maps of these factors will help identify areas within the system where relative attenuation is high or low throughout the region. The relative attenuation of a stream or river reach will be considered attenuation on a per kilometer basis for comparative purposes in this report.

Watershed Scale Attenuation of Rivers and Streams in Puget Sound Basin

At the watershed scale, two empirical attenuation models were used to estimate average relative annual attenuation in the Puget Sound Basin for the networks of the 17 major rivers draining to Puget Sound (fig. 1). The river nitrogen (RivR-N) model (Seitzinger and others, 2002), which is based on only physical (hydrological) characteristics and the v_f model, which incorporates physical, chemical, and biological factors, were used (Doyle, 2005; Wollheim and others, 2006).

Table 2. Summary of modeling approaches to describe instream nutrient attenuation for watershed-scale models from review of scientific literature.

Attenuation expression	Nutrient	Parameter categories			Description	References
		Physical	Chemical	Biological		
$R = Sn / ((z / T) + Sn)$ where $Sn = f / [NO_3^-]$	Nitrate	X	X	X	This method uses physical characteristics of the stream and information on nutrient concentration and denitrification rates. Initially developed for lakes, but applicable for any unstratified water body. Where R is instream attenuation defined as the percentage of nutrient input removed, z is mean depth, T is residence time, Sn is mass transfer coefficient, f is the measured denitrification rate and $[NO_3^-]$ is the nitrate concentration.	Kelly and others (1987); Howarth and others (1996)
$y = 10^{(1.00 \log (x/Q) - 0.39)} / Q$	Total nitrogen (TN)	X	X		Based on previous work showing that instream attenuation (y) increases with loading rates (x), a regression was developed for discharge (Q) normalized data form lakes, wetlands, and rivers. Mass balance data (outputs – inputs) from the literature were used. y = nitrogen attenuation in grams per square meters per year and x is nitrogen load in grams per square meters per year.	Saunders and Klaff (2001)
$R = 88.45(D / T)^{-0.3677}$	Nitrate	X			A regression model, the RivR-N model, was used to simulate the proportion of nitrate removed from streams and reservoirs based on published nitrate removal data. Data were compiled and quantified from mass balances (upstream-downstream loads) or denitrification studies in sediment cores. R = nutrient attenuation as proportion of nitrate inputs removed; D = mean depth in meters; T = time of travel in years. T was calculated by taking reach length divided by the velocity. D was calculated from mean annual discharge (Q) as $D = 26.12Q^{-0.3966}$ (Leopold and Maddock, 1953).	Seitzinger and others (2002)

Table 2. Summary of modeling approaches to describe instream nutrient attenuation for watershed-scale models from review of scientific literature.—Continued

Attenuation expression	Nutrient	Parameter categories			Description	References
		Physical	Chemical	Biological		
$R = 1 - \exp(-LVf / uh); 1 - \exp(-Vf / H_L)$	Nitrogen or phosphorus	X	X	X	Retention of nutrient as a part of incoming load (R) based on an analysis of the hydrologic variability on nutrient spiralling. Beginning with assumptions of nutrient sprialing, first order exponential decay along reach and low influence of dispersion and transient storage, this expression for R, takes into account the hydrological and biological characteristics of the reach. v_f= uptake velocity for nitrogen or phosphorus; L = reach length; u = mean velocity; h = mean depth; H_L is the hydraulic load defined as Q/wL = h/T = uh/L, where Q is discharge, w is width and T is residence time.	Doyle (2005); Wollheim and others (2006); Mulholland and others (2008); Helton and others (2010); Stewart and others (2011); Aguilera and others (2013)
$k1 = \exp(-\delta T); k2 = \exp(-\alpha 1 D \alpha 2 T)$	Nitrogen or phosphorus	X	X	X	The SPARROW model is a hybrid statistical and mechanistic model for estimating the movement of mass through the landscape under long-term, steady state conditions. A calibrated model provides estimates of the annual load and annual yield (load per unit area) for the constituent of interest, for example TN and total phosphorus (TP), for every reach in the modeled surface-water network. The instream decay (k1) is a function of δ, the decay coefficient (fitted with the model), and T is the travel time in the reach. In later uses, the expression given by k2 and was used where α1 and α2 are coefficients fitted by the model, D is the reach depth.	Smith and others (1997); Alexander and others (2000); Alexander and others (2007)
$PHI = (par2)(C_i)(T_{10})(V)$	Nitrogen	X	X	X	The hydrologic model (HBV) uses daily precipitation and monthly evaporation data to simulate water flow in the river network. The watershed model is calibrated to a time series for outlet discharge. Instream attenuation (PHI) is a function of par2, a fitted parameter, which is site specific, C_i is the inorganic nitrogen concentration, T_{10} is average of the last 10 day air temperature, and V water volume in storage.	Arheimer and Brandt (1998)

Table 2. Summary of modeling approaches to describe instream nutrient attenuation for watershed-scale models from review of scientific literature.—Continued

Attenuation expression	Nutrient	Parameter categories			Description	References
		Physical	Chemical	Biological		
$R = K \times \left(10^{0.0293T} \times Ab\right) / Q$	Nitrogen	X	X	X	HYDRA is a riverine flow and transport model, and authors assumed NO_3^- removal was primarily though denitrification and associated with the sediments. For each grid cell of the model defined R, the fraction of nitrogen removed where T is the water temperature, K is a rate parameter (m/sec) fitted with the model; Ab is the river bed area (m^2), Cn is NO_3^- concentration (kg/m^3).	Donner and others (2004)
$R = 1/\left(x1\left(1000slope+1\right)q^{x2}\right)$	Nitrogen	X			The POLFLOW model, a GIS-based model that simulates water flow, nutrient transport, and attenuation at a 1×1 km^2 GIS pixel scale. The expression for R, the fraction of nitrogen removed in each pixel is based on ×1 and ×2, which are fitted model parameters, slope is the topographic slope of each cell, and q is average water flow for each pixel.	Darracq and Destouni (2005)
$k = 0.14\ln\left(Tw\right) - 0.28$	Nitrogen			X	This modeling approach estimated nitrogen retention in streams and rivers using a longitudinal transport model with storage and variable residence time (VART), and incorporating a first-order reaction term. Monthly mean discharge and water-quality data and watershed modeling using the hydrologic simulation program Fortran (HSPF) model to parameterize the model. The instream nutrient loss (k) of the model is a function of Tw, the monthly mean water temperature and based on site-specific correlations.	Jung and Deng (2011)

RivR-N Attenuation Model

The RivR-N model (Seitzinger and others, 2002) is a predictive model relating the physical and hydraulic properties of rivers and lakes to the proportion of N input that is removed by the water body. Published data on N removal from river reaches from the 1970s through 1990s were compiled that ranged from first order headwater rivers to the mouths of major rivers in the United States, Europe, New Zealand, and Canada. Land use in the watersheds included agricultural, urban, forested, and mixed uses. The proportion of N removal was quantified from mass balances (changes between upstream-downstream loads) or denitrification studies in sediment cores. Of the mass balance studies used, most were based on mass balance of NO_3^-, or in cases of mass balances on total N, loss of total N in each study was likely from denitrification. As a result, the RivR-N model primarily represents annual rates of NO_3^- removal (by denitrification) across various rivers as the original study was focused on watershed N losses (Seitzinger and others, 2002). The relations between removal and river order, river discharge, watershed land use, total N-loading to the watershed, surface water residence time, and the water displacement ratio were examined. The best model, derived from 33 studies, was based on the ratio of depth to time of travel and given by:

$$R = 88.45(D/T)^{-0.3677} \tag{4}$$

where

- R is nitrate removal as a fraction of total reach inputs,
- D is the average reach depth in meters, and
- T is time of travel in the reach in years.

The time of travel is typically calculated by taking the reach length (L) divided by the mean velocity (u) of the reach. The displacement ratio (D/T) represents the height of water column annually displaced from the water body (Seitzinger and others, 2002). This model provides a measure of the extent to which nutrients are in contact with the benthic sediments and a measure of variations in water contact time that result from changes in channel size (velocity and depth). Although the model was not developed with data from the PNW, and did not include rivers that are glacially influenced, the model has been successfully used to describe patterns of instream attenuation across 16 watersheds in the eastern United States (Seitzinger and others, 2002). Because the model describes a relation between N (as NO_3^-) removal from a wide variety of studies, it was applied to the Puget Sound lowlands for this first-level analysis because estimates of channel characteristics in equation 4 were available.

The RivR-N model was applied to a hydrologic network consisting of 535 valley segments (Konrad, 2015) that are based on the Puget Sound Watershed Characterization (Washington Department of Ecology, 2013b) (fig. 1). This river network is based on the high-resolution (1:24,000) flow lines from the National Hydrography Dataset (NHD) for Washington (U.S. Geological Survey, 2013). Flow lines for the 17 major rivers flowing into Puget Sound were clipped, and any segments draining less than 50 km^2 were removed. Therefore, analysis of relative attenuation presented in this report do not include the smallest headwater streams for these 17 major river basins.

The D/T in equation 4 is also referred to as the hydraulic load (H_L) (Wollheim and others, 2006) and can be defined in terms of streamflow and the surface area of the streambed by D/T = Du/L = Q/wL, where w is the reach width and Q (= wDu) is the streamflow of the reach, and u is the mean velocity and L is the reach length. Therefore, the RivR-N model requires reach-scale estimates of Q/wL to estimate attenuation within stream and river reaches the Puget Sound Basin. Mean annual discharge for each of the 535 river segments was determined from an area-discharge relation derived from historical streamflow data for western Washington collected by U.S. Geological Survey (USGS). Streamflow data were downloaded from the USGS National Water Information System (NWIS) database (http://waterdata.usgs.gov/wa/nwis/nwis) and mean annual discharge was calculated at all sites with at least 5 years of daily streamflow measurements for the 1981–2010 period (current National Oceanic and Atmospheric Association climate normal period). The drainage area for these streamgages was also compiled from NWIS site files and an area-based regression for mean annual streamflow based on 147 USGS streamgages in western Washington with drainage areas greater than 10 square kilometers was created (fig. 6; appendix A). Drainage areas for the river segments in the hydrologic model are provided in Konrad (2015); the equation in figure 6 was used to estimate mean annual flow in all 535 river segments.

Width in meters was derived using regression equation 5, which describes the hydraulic geometry for rivers in Washington State (Magirl and Olsen, 2009):

$$w = 4.85 * Qmean^{0.48}/3.281 \tag{5}$$

where

- w is mean annual width in meters,
- Qmean is an estimate of the mean annual flow in cubic feet per second, and
- 3.281 is a conversion factor to convert width from feet to meters.

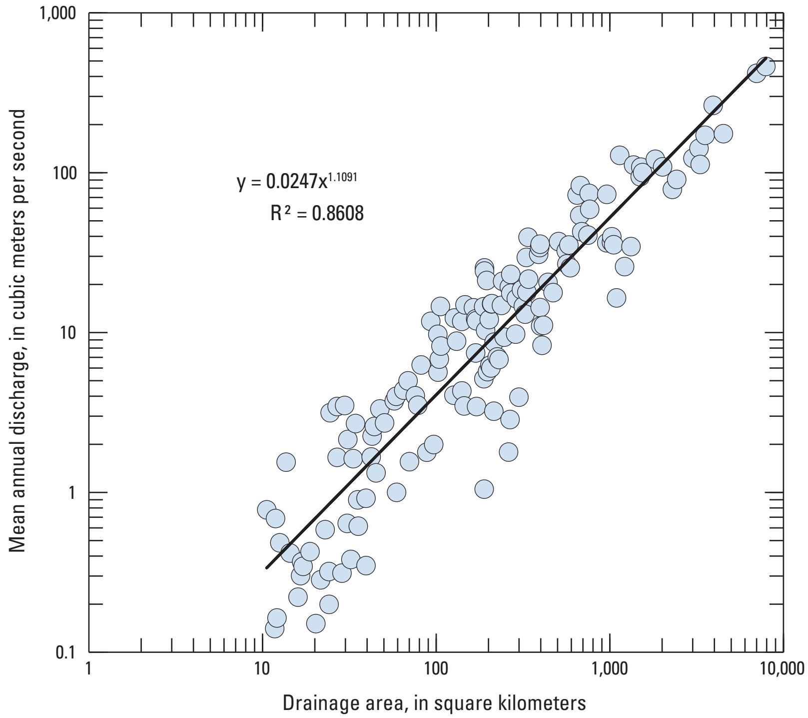
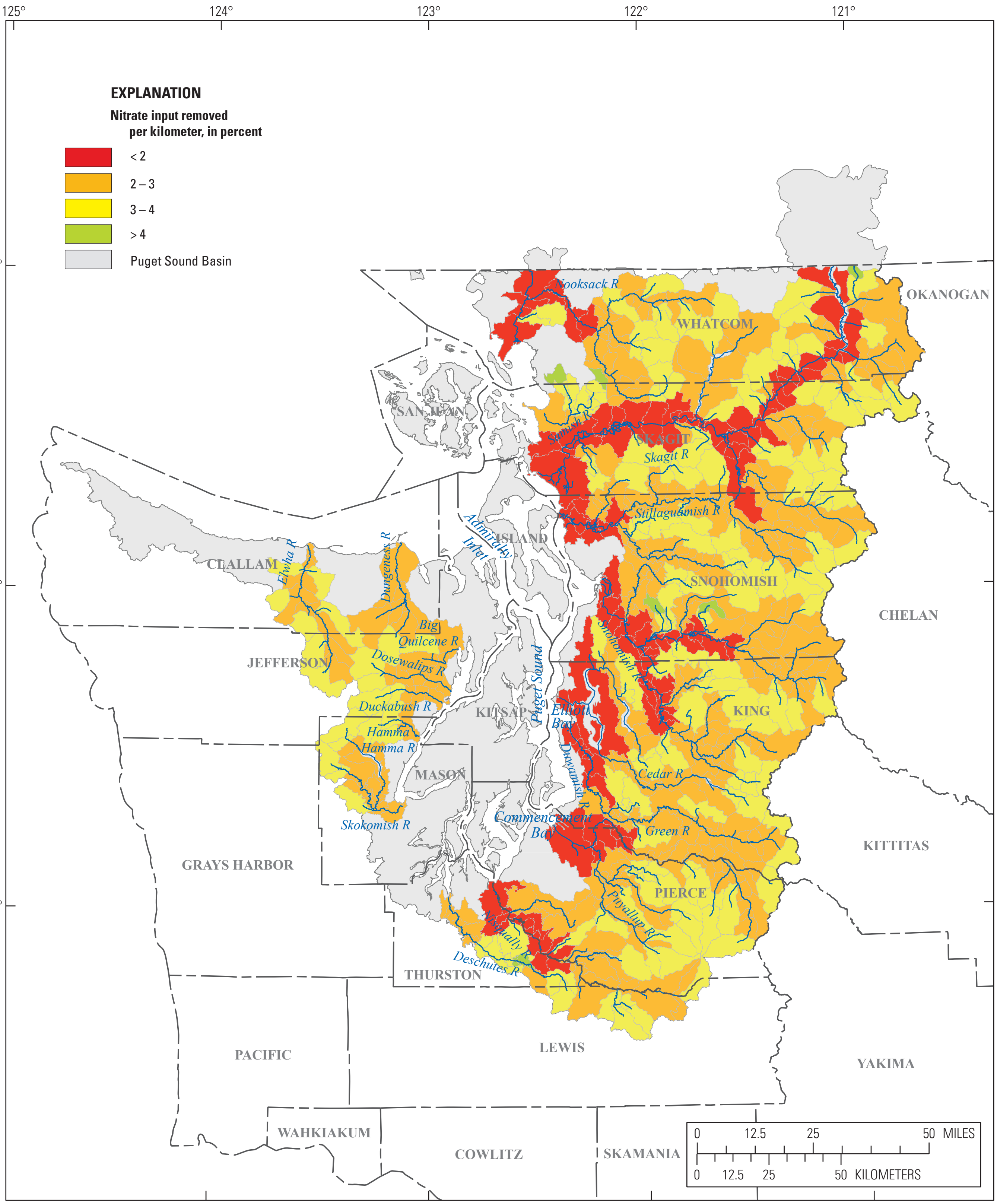


Figure 6. Mean annual discharge and drainage area for 147 U.S. Geological Survey streamgages with at least 5 years of record, western Washington, 1981–2010.

Equation 5 is based on data from USGS streamgages that are located on areas of river reaches where channels are well defined; therefore, for unconfined reaches, or braided channels with complex geomorphologies these width estimates will underestimate the actual channel widths. As a result, NO_3^- attenuation (R) will be underestimated for these reaches as well. Lastly, a standardized reach length (L) of 1,000 meters was selected to calculate a reach normalized H_L. Because a standardized reach length was used, all attenuation values calculated for Puget Sound in this report are referred to as "relative attenuation." This approach was selected to normalize attenuation across all reaches so that areas with low and high attenuation potential could be highlighted. In order to display attenuation data more clearly on a map, the relative instream NO_3^- attenuation values for each reach are shown spatially by applying these values to their incremental contributing drainage area and do not represent attenuation within the watershed (fig. 7).

Generally, the reaches with the highest relative attenuation (>4 percent of NO_3^- inputs attenuated per kilometer) were few (only 8 of the 535 river segments) and located in a handful of headwater reaches scattered throughout the Puget Sound Basin. Most reaches showed moderate levels of attenuation (ranging from 2 to 4 percent of NO_3^- inputs attenuated per kilometer). The lowest relative attenuation was in the main stems of the largest of the 17 major rivers ranging from the Nooksack River in the north to the Nisqually River in the south (< 2 percent of N inputs attenuated per kilometer; fig. 7). The channels with the lowest relative attenuation tended to be wide and deep, which resulted in a lower displacement ratio and lower attenuation. Low relative attenuation was calculated throughout the entire main stem of the Skagit River from Canada to where it enters Puget Sound (fig. 7). For each of the major river basins, the RivR-N model shows higher relative attenuation in smaller headwater reaches than in their respective main stem reaches. These results do not necessarily mean that attenuation is not important in the main stems, but rather the RivR-N model shows that main stems are not as efficient at attenuating NO_3^- as the smaller headwater reaches are.

Overall, the fraction of NO_3^- inputs attenuated per kilometer of stream length using the RivR-N model ranged from 1.3 to 4.6 percent with an overall mean of 2.7 percent (table 3). The distribution of RivR-N attenuation showed only a few reaches with NO_3^- inputs attenuated per kilometer greater than 4 percent (fig. 8), and a majority of the data was centered around the mean.



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 7. Relative instream nitrogen attenuation expressed in percentage of nitrate inputs removed per kilometer of stream length using the RivR-N model for Puget Sound Basin, Washington. The instream attenuation values for each reach are shown spatially by applying these values to their incremental contributing drainage area and do not represent watershed attenuation. Gray areas represent areas of the Puget Sound lowlands that do not include a major river.

Table 3. Summary statistics of modeled concentrations, v_f values, and reach-level relative attenuation from the RivR-N model and v_f models expressed as a percentage of nutrient input attenuated per kilometer of stream length, Puget Sound Basin, Washington.

[Relative attenuation is defined as the percentage of nutrient inputs removed per kilometer of stream reach. **Abbreviations:** mg/L, milligram per liter; mm/min, millimeter per minute]

	Nitrate (mg/L)	Phosphate (mg/L)	Nitrate v_f (mm/min)	Phosphate v_f (mm/min)	Attenuation model (percent)		
					RivR-N	v_f Nitrate	v_f Phosphate
Minimum	0.0	0.0	0.02	0.15	1.3	0.03	0.23
Maximum	7.1	1.5	2.2	0.50	4.6	32	8.5
Mean	0.25	0.02	0.99	0.41	2.7	4.8	2.0
Median	0.11	0.01	0.93	0.41	2.7	4.0	1.8
Sample N[1]	11,213	11,223	492	489	517	492	489

[1]Nitrate and orthophosphate concentration data were calculated from the output from the PNW SPARROW model clipped to the Puget Sound Basin. The data for proportion of nutrient attenuation and v_f are based on the 535 reaches of the aggregated hydrologic model.

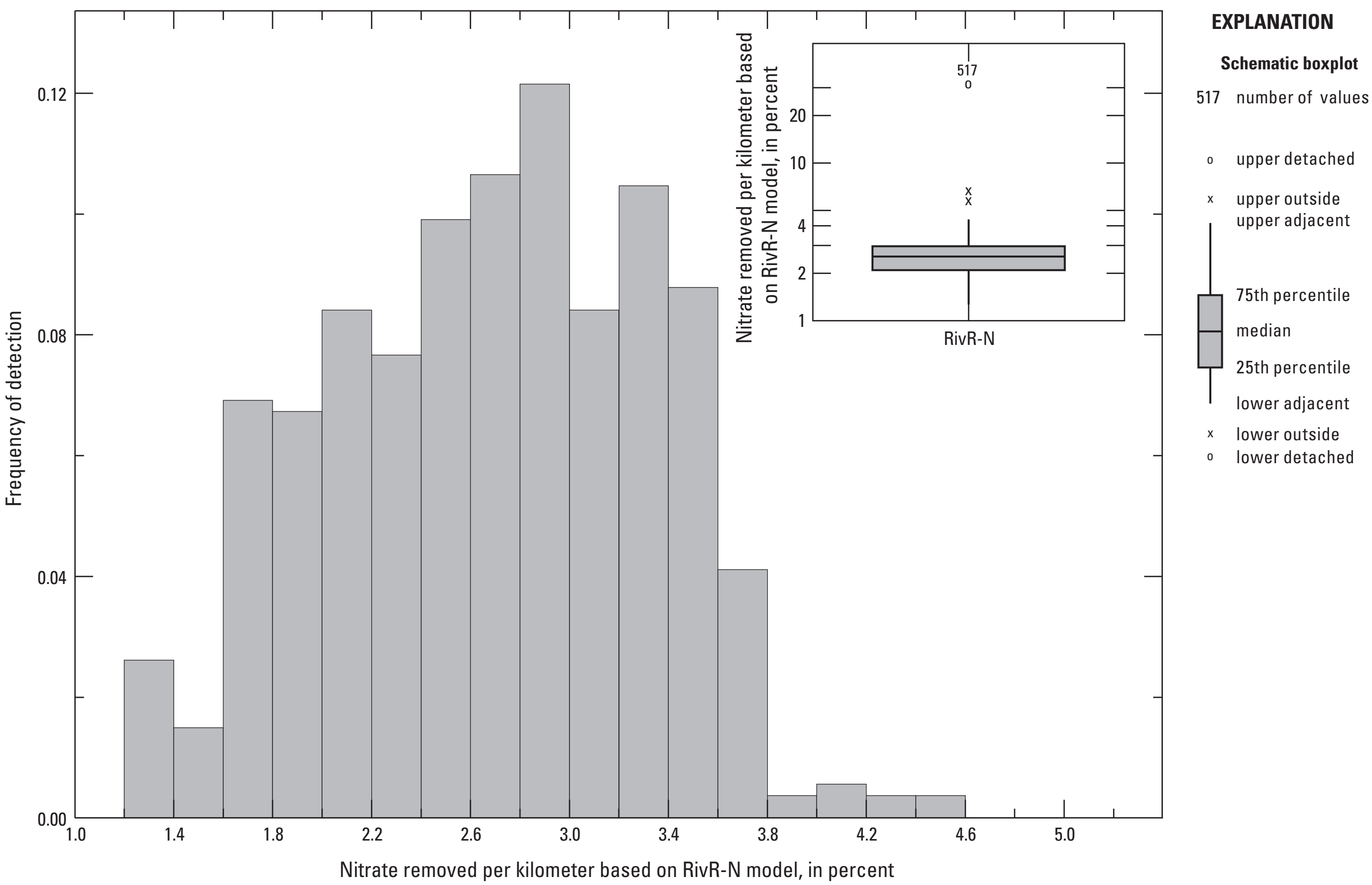


Figure 8. Relative instream attenuation calculated using the RivR-N model expressed as the percentage of nitrate removed per kilometer of stream length, Puget Sound Basin, Washington.

v_f Attenuation Model

The second watershed attenuation model used was the v_f model. This model was first developed as a generic expression for describing the effects of hydrologic variability on nutrient attenuation measured from nutrient spiraling metrics (Doyle, 2005) and has been used by several researchers for describing attenuation in watersheds throughout the United States and Spain (Wollheim and others, 2006; Mulholland and others, 2008; Helton and others, 2011; Aguilera and others, 2013). The v_f model defines attenuation (the same as the RivR-N model) as the amount of nutrients removed as a fraction of total reach inputs (R) by:

$$R = 1 - \exp(-v_f/H_L) \qquad (6)$$

where

H_L is the hydraulic load and
v_f is the nutrient uptake velocity.

The uptake velocity is equivalent to a mass transfer coefficient and represents how efficient a given reach is at taking up (retaining) nutrients. As v_f increases, the capacity of a stream to retain nutrients also increases. Furthermore, v_f is independent of river size in benthic dominated systems (Wollheim and others, 2006) making it an ideal biological parameter for estimating nutrient attenuation. As a result, the v_f model incorporates biological factors of nutrient attenuation through v_f and the hydrologic condition of the stream channel through H_L. The v_f model was derived beginning with two assumptions about nutrient spiraling: attenuation follows first order exponential decay along a reach, and nutrient transport is minimally influenced by dispersion and transient storage and is a generic expression that can be applied to any watershed.

The v_f model for attenuation was applied to the same hydrologic network described for the RivR-N model above and H_L was calculated the same way. To parameterize the v_f model fully, segment-scale estimates of v_f are needed for each nutrient being studied. To date (2015), there are no known estimates of v_f for streams and rivers in the Puget Sound Basin. However, Mulholland and others (2008) calculated NO_3^- uptake across 72 streams in the United States and showed that v_f for denitrification was a function of stream NO_3^- concentration (range was 0.001 to 21.2 milligrams of nitrogen per liter [mg N/L]). The general expression relating v_f to nutrient concentration (Helton and others, 2011) is:

$$v_f = aC^b \qquad (7)$$

where

C is nutrient concentration (either NO_3^- or (PO_4^{-3}), and
a and b are fitted by plotting v_f values calculated from field experiments to surface water concentrations during the time of each field experiment and these coefficients are dependent on study area.

The expression given by equation 7 allows for "biologic saturation" of nutrient attenuation at high concentration values, which has been shown to occur in various small stream types (pool/riffle, channelized reaches, reference, agricultural, and urban land uses) (Dodds and others, 2002; Mulholland and others, 2008).

Aguilera and others (2013) derived equations for v_f for NO_3^- and PO_4^{-3} based on a review of the scientific literature. In that study, the authors compiled data for measured v_f values and concentrations for NO_3^- and PO_4^{-3} from a wide variety of small streams across the United States located in reference, agricultural, and urban watersheds, and focused on studies that used stable isotopes or mass balances in order to focus on net retention of nutrients. Aguilera and others (2013) purposely omitted short-term nutrient addition studies because they tend to focus on gross attenuation, not net attenuation. Their review of the scientific literature included more than 100 studies of NO_3^- retention (concentration range from 0.001 to greater than 10 mg N/L) and more than 60 studies of PO_4^{-3} retention (concentration range from 0.001 to greater than 10 milligrams of phosphorus per liter [mg P/L]) to derive the following relations between v_f and nutrient concentration:

$$v_f^N = 0.41[NO_3^-]^{-0.39} \qquad (8)$$

$$v_f^P = 0.25[PO_4^-]^{-0.11} \qquad (9)$$

where

v_f is in units of millimeters per minute and concentration in units of milligrams per liter.

No data from Puget Sound streams were included in the formulation of equations 8 and 9; however, several sites in western Oregon that ranged from pristine forested reaches to highly urbanized reaches were included. Because the range in concentrations represented in equations 8 and 9 encompasses the values of nutrients measured in Puget Sound (Embrey and Inkpen, 1998; Ebbert and other, 2000), we determined that for this first level analysis they are applicable to Puget Sound streams as well. Some differences from these equations and Puget Sound sites are expected in larger rivers where interactions between groundwater and surface water might be minimal. Equations 8 and 9 were used to estimate segment-scale v_f values for each reach in the Puget Sound hydrologic network through a multistep process.

Using the results from a recently published total N (TN) and total P (TP) SPARROW model for the PNW, specific results for mean annual TN and TP loads were extracted from Wise and Johnson (2013) output files (Daniel Wise, U.S. Geological Survey, written commun., 2014) for the 535 river segments of the Puget Sound hydrologic network. SPARROW is a watershed statistical model that uses comprehensive input datasets for water quality, watershed characteristics, and total watershed (terrestrial) nutrient inputs to simulate instream nutrient loads for each segment of a model river network (Smith and others, 1997). For the PNW SPARROW model, the river network was defined using the NHD (Horizon Systems Corporation, 2012) and the mean annual discharge and reach-scale TN or TP load from SPARROW output was compiled for the 535 segments in the river network. These mean annual TN and TP loads then were converted into mean annual TN and TP concentrations by dividing the loads by the mean annual discharge from the NHD for each modeled reach. The discharge values given by the NHD are long-term mean annual flow estimates adjusted to long-term (1971–2000) mean annual discharge values measured at USGS streamgages. Therefore, the discharge values represent the average values between 1971 and 2000 (Horizon Systems Corporation, 2012). The NHD discharge values were selected instead of discharge values from the area-flow regression used previously (fig. 6) because the NHD data were used to model the nutrient load and the same discharge values were needed to convert from load to concentration for consistency with how these data were produced. Reach-scale estimates of TN and TP concentrations were converted to NO_3^- and PO_4^{-3} concentrations using simple linear regressions between TN and NO_3^- and TP and PO_4^{-3} concentrations from historical water-quality data for Puget Sound available from the USGS NWIS database. The regression equations were developed with the intercept set at zero. A statistically significant relation between TN and NO_3^- from the USGS NWIS database was developed for 197 measurements for streams in the Puget Sound Basin between 2003 and 2011. Those observations with values less than the detection limit of 0.01 mg N/L were not used in the analysis. The N analysis resulted in the following model: NO_3^-, mg/L = 0.76 (total N, mg/L) ($R^2 = 0.96$; $p<0.0001$) (fig. 9*A*). A relation between TP and ortho-P was also developed using data from the NWIS database. The P model used 322 observations and, like the N model, data less than the detection limits (typically 0.01 mg P/L) were excluded. Sites in the database that were tidally influenced or that were dramatically altered hydraulically (that is, diversion canals, outlets from dams) were eliminated from the analysis because of the way these systems influence sediment and therefore sediment bound P. The P analysis resulted in the following model: PO_4^{-3}, mg/L = 0.40 (total P, mg/L) ($R^2 = 0.76$; $p<0.0001$) (fig. 9*B*). The P analysis did exhibit heteroscedasticity (fig. 9*B*), and although this can result in a biased estimate of the standard error for a coefficient, it does not result in a biased estimate of the coefficient itself. This limitation was viewed as acceptable for the exploratory and hypothetical analysis done for this study. After these reach-level estimates of mean annual NO_3^- and PO_4^{-3} concentrations were calculated, equations 8 and 9 were used to calculate the respective v_f value and the relative attenuation value (equation 6) for each reach in the Puget Sound hydrologic network.

Summary statistics of the modeled nutrient concentrations from SPARROW used to calculate NO_3^- and PO_4^{-3} v_f values are given in table 3 and are within the range of observed values for Puget Sound (Embrey and Inkpen, 1998). This indicates that the method for estimating instream concentrations of NO_3^- and PO_4^{-3} is producing reasonable concentration data. Values of v_f calculated for NO_3^- ranged from 0.02 to 2.2 mm/min and for PO_4^{-3} 0.15 to 0.50 mm/min (table 3); both ranges were within the ranges found in the literature, but tended to be on the lower end of those ranges (Ensign and Doyle, 2006; Aguilera and others, 2013).

It should be noted that the coefficients used to describe attenuation in the PNW SPARROW model were not significant explanatory variables of nutrient load (Wise and Johnson, 2013). These results should not be used to assume attenuation is not occurring in Puget Sound streams and rivers. Rather, it is possible that the available input data or the expression used to characterize attenuation in the SPARROW model were temporally, spatially, or conceptually inadequate to characterize this process. Additionally, the SPARROW model is not optimized for Puget Sound, but rather the entire PNW, so although attenuation was not significant across this entire region, it still may be a factor within the smaller Puget Sound region.

Results from the v_f model of attenuation for NO_3^- had some features similar to the RivR-N model for NO_3^-, but also exhibited some differences (figs. 7 and 10). For NO_3^-, the highest relative attenuation per kilometer of stream length (>4 percent of nitrate inputs attenuated) was in the headwaters of almost all the major rivers. The main stems and mouths of most major rivers consistently showed the least amount of relative attenuation (< 2 percent of NO_3^- inputs attenuated, fig. 10) throughout the Puget Sound Basin. The major rivers of the Olympic Peninsula in the western Puget Sound Basin, also show higher relative attenuation throughout their watersheds than basins draining the Cascade Range in the eastern Puget Sound Basin. Watersheds on the Olympic Peninsula tend to have lower nutrient concentrations, lower mean annual discharge, and wider shallow channels than more populous basins that tend to have higher nutrient loads and channel simplifications from developed land uses. The relative NO_3^- attenuation per kilometer ranged from 0.03 to 31.8 percent with a median of 2.7 percent, and the distribution showed that the maximum relative attenuation value was an anomaly (fig. 11). Without this one high value, the range of relative NO_3^- attenuation ranged from 0.03 to 15.5 percent of inputs to the reach per kilometer.

For PO_4^{-3}, the patterns of relative attenuation per kilometer of stream length are similar to patterns for NO_3^-. Relative attenuation tended to be high in headwater reaches and low in the main stem and mouths of the major rivers (fig. 12). Only a few reaches (15 of 489) show relative attenuation greater than 4 percent per kilometer of stream reach. Reaches having the lowest relative attenuation are (< 2 percent of inputs removed per kilometer) much more pervasive for PO_4^{-3} than for NO_3^-. This might be an artifact of the v_f model for PO_4^{-3}, which is based on a biologic removal of P and might not fully incorporate P attenuation related to sedimentation of P bound to suspended sediment.

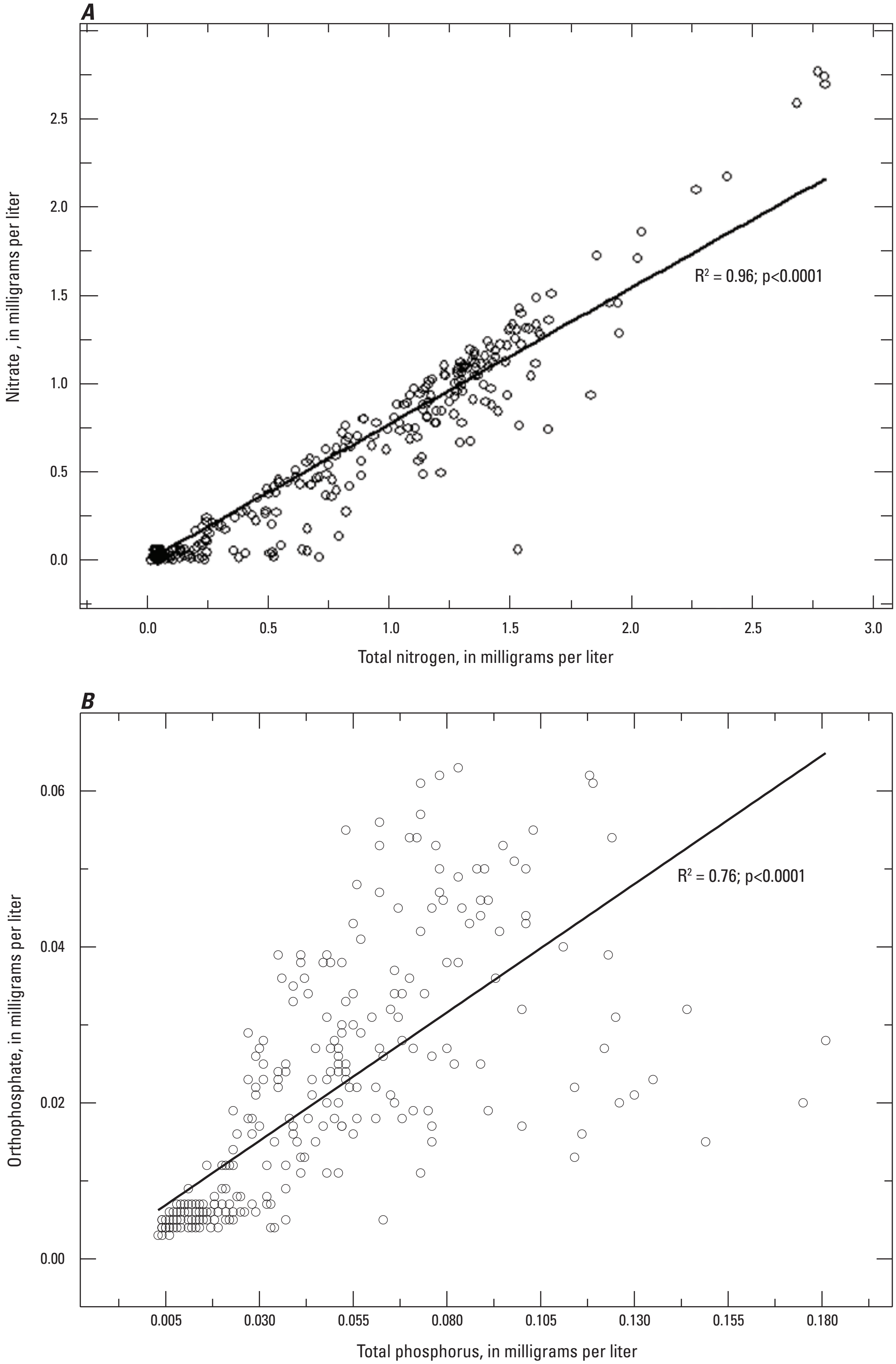
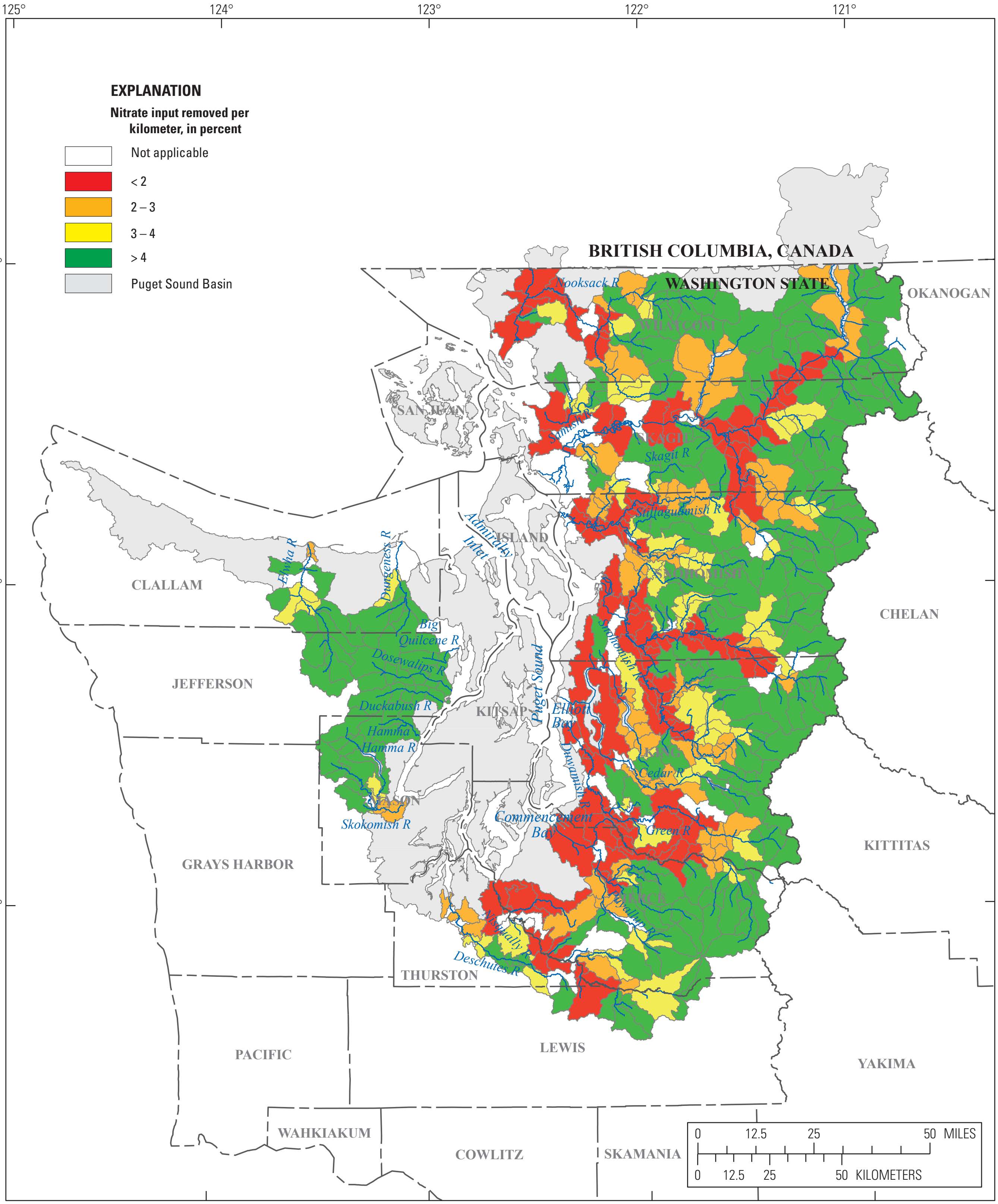


Figure 9. Relation between total and dissolved nutrient concentrations for (*A*) total nitrogen and nitrate and (*B*) total phosphorus and orthophosphate for data compiled from the U.S. Geological Survey National Water Information System, Puget Sound Basin, Washington.



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 10. Relative instream attenuation expressed as the percentage of nitrate inputs removed per kilometer of stream length based on the v_f model, Puget Sound Basin, Washington. The relative instream attenuation values for each reach are shown spatially by applying these values to their incremental contributing drainage area and do not represent watershed attenuation. Gray areas represent areas of the Puget Sound lowlands that do not include a major river. White areas are reaches where a nitrate concentration could not be calculated.

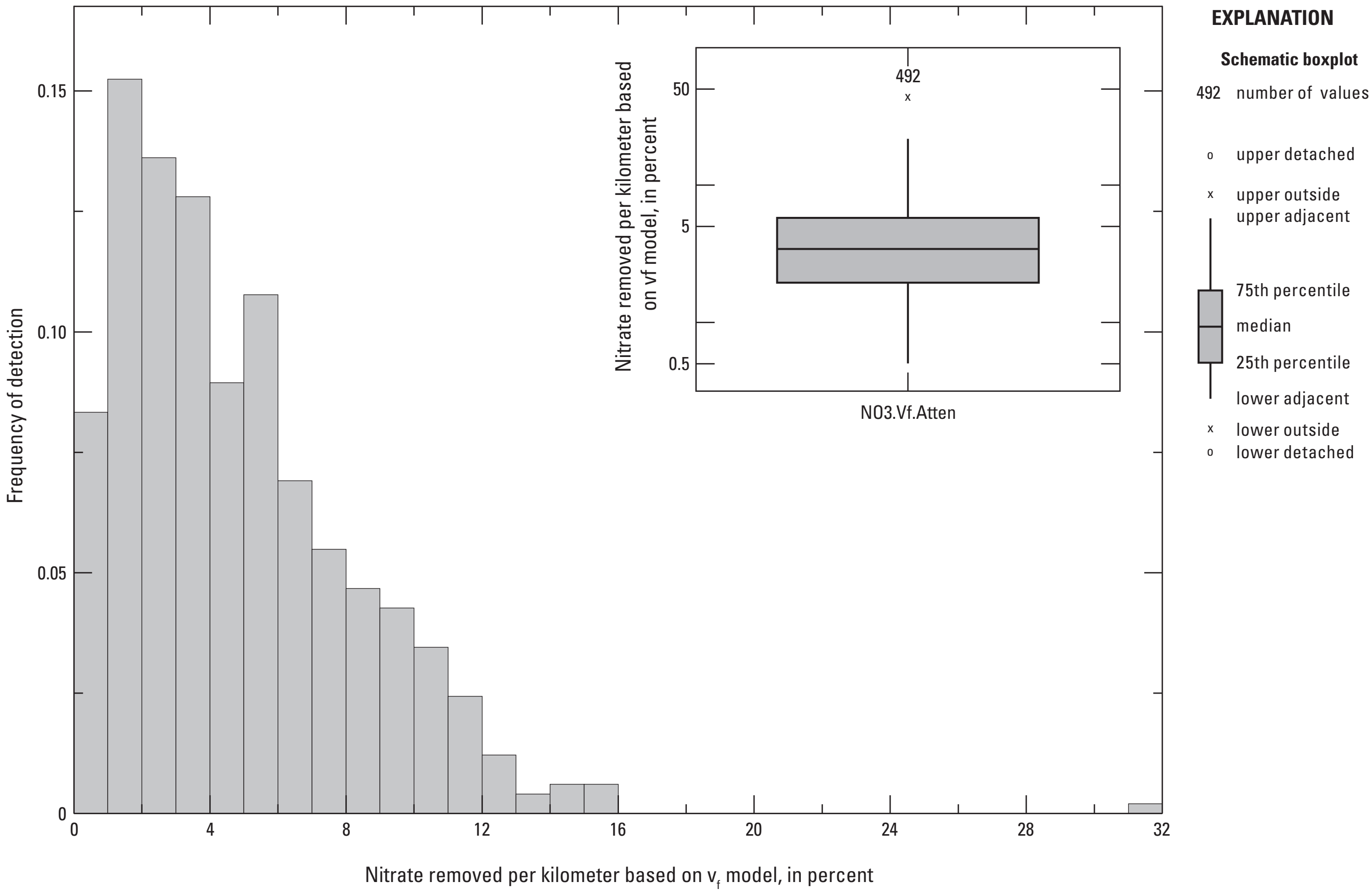
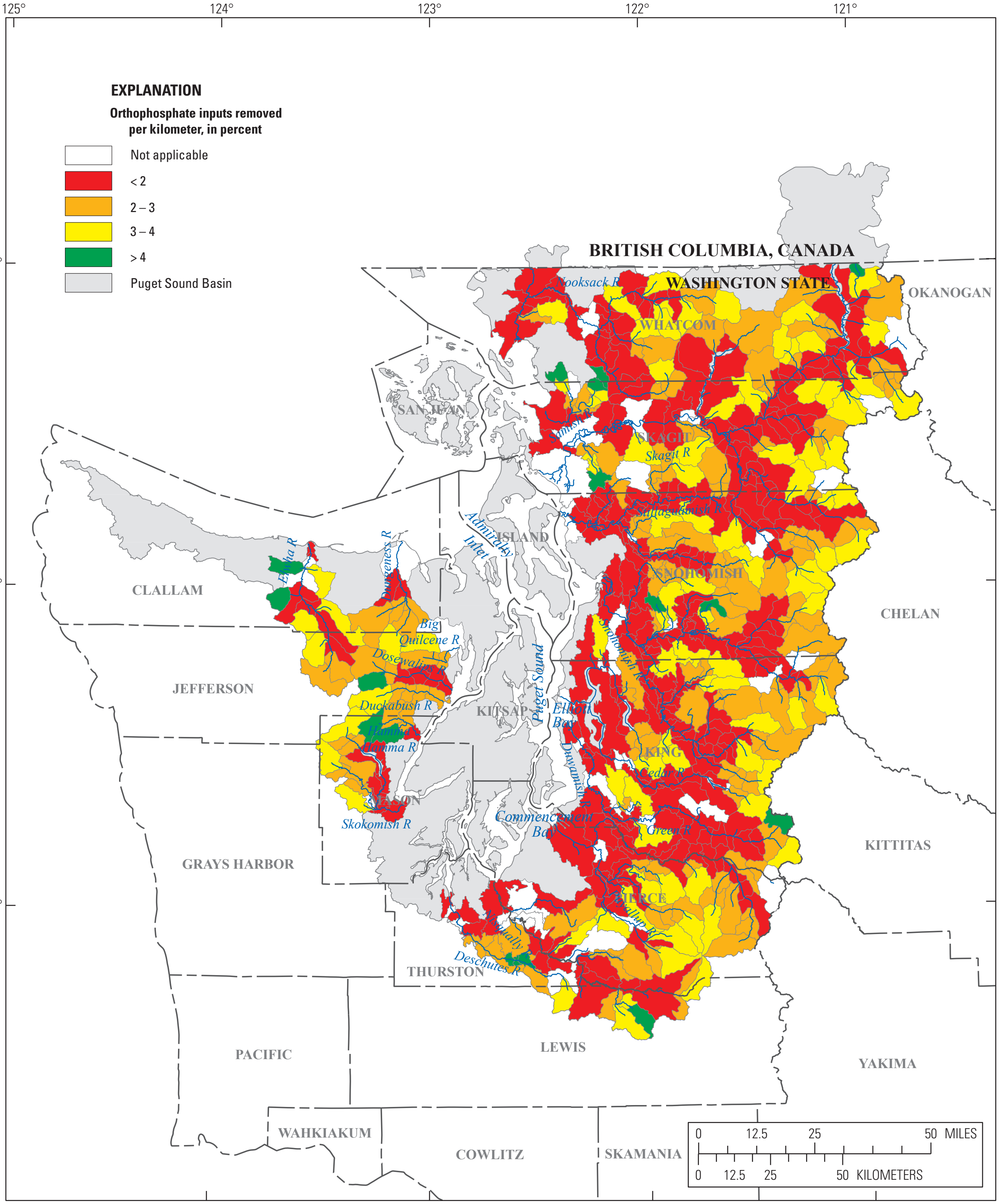


Figure 11. Relative instream attenuation based on the fraction of nitrate removed per kilometer of stream length using the v_f model, Puget Sound Basin, Washington.



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 12. Relative orthophosphate attenuation represented by the percentage of orthophosphate inputs removed per kilometer of stream length based on the v_f model, Puget Sound Basin, Washington. The instream orthophosphate attenuation values for each reach are shown spatially by applying these values to their incremental contributing drainage area and do not represent watershed attenuation. Gray areas represent areas of the Puget Sound lowlands that do not include a major river. White areas (values not applicable) are reaches where an orthophosphate concentration could be calculated.

Additionally, the range of v_f values calculated for PO_4^{-3} (0.15 to 0.50 mm/min; table 3) is small and is on the low end of the range in values summarized elsewhere (Ensign and Doyle, 2006; Aguilera and others, 2013) where v_f for PO_4^{-3} can be greater than 6 mm/min. The distribution of the PO_4^{-3} attenuation was skewed toward the low end with a few high values, but most reaches showed less than 3 percent of inputs attenuated per kilometer (fig. 13).

Note that the attenuation categories presented in figures 7, 10, and 12 were selected in order to be consistent across these figures and to facilitate comparisons between the two models and nutrient species (NO_3^- and PO_4^{-3}) being considered. The main reason for these analyses is to examine patterns of relative attenuation across the Puget Sound Basin and not to focus on the absolute values of reach-scale relative attenuation. These categories capture the range of data adequately; however, for relative NO_3^- attenuation from the v_f model, numerous reaches are greater than 4 percent of inputs attenuated per kilometer (fig. 11) and should be noted. Furthermore, it is important to note that the values presented represent reach-level attenuation per kilometer and not the cumulative attenuation of the entire river basin, which would be greater.

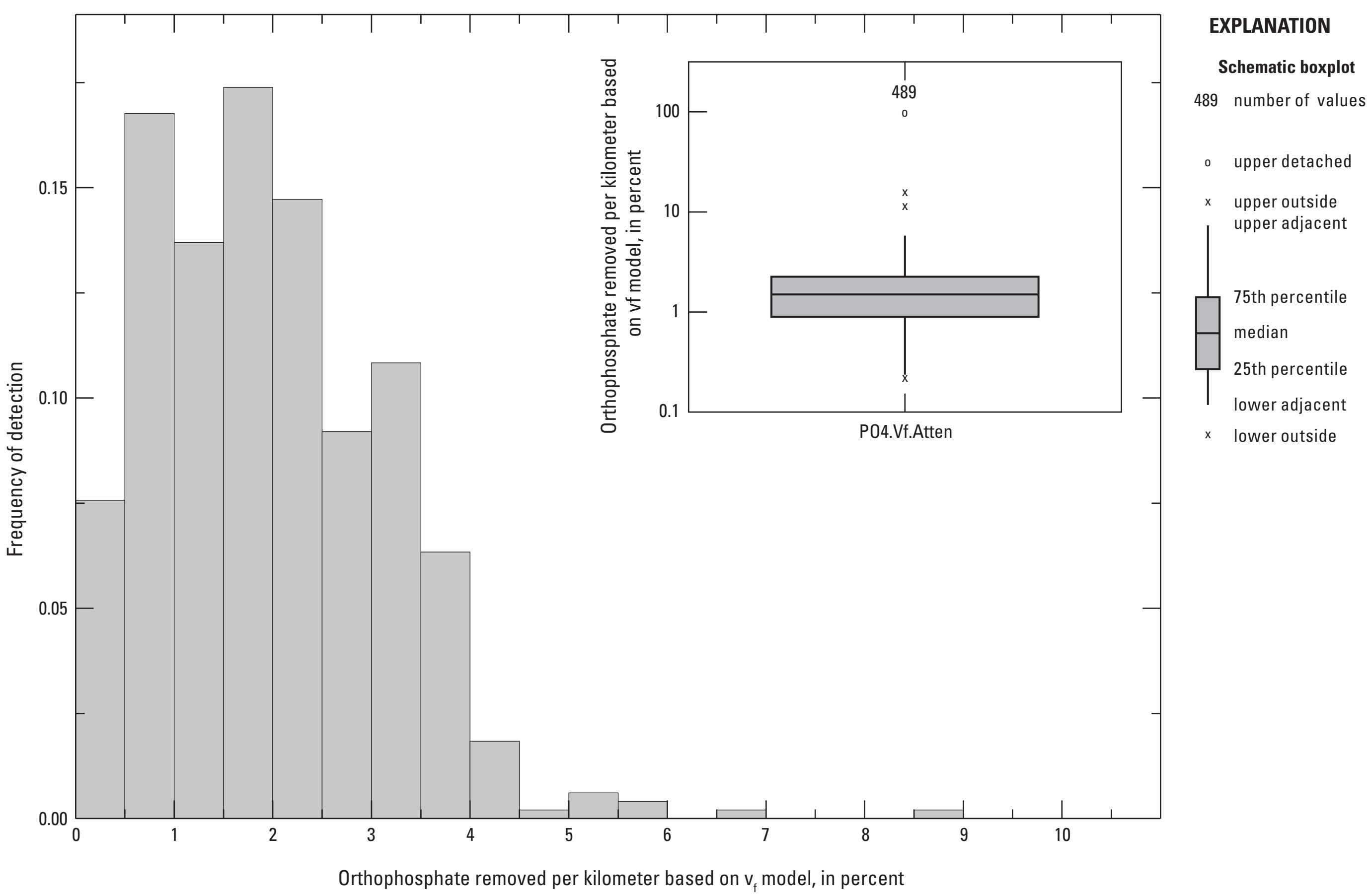


Figure 13. Calculated relative instream attenuation based on the fraction of orthophosphate removed per kilometer of stream length using the v_f model, Puget Sound Basin, Washington.

Comparison of Relative Nitrate Attenuation Models

Two approaches were used for estimating relative NO_3^- attenuation for the Puget Sound Basin: the RivR-N and v_f models. Both of these models estimate NO_3^- loss from similar processes, namely denitrification, and a comparison of the output for these two models is provided in figure 14. Positive values in figure 14 indicate that more attenuation is being predicted by the NO_3^- v_f model and this comparison shows that in almost all cases, relative NO_3^- attenuation is greater when using the v_f model. These differences are highest in headwater reaches for all the major river basins. However, in many of the lower main stem reaches, relative NO_3^- attenuation from RivR-N model is greater, as indicated by the negative values. The RivR-N model is based solely on physical (hydrological) characteristics of a given stream or river reach, whereas the v_f model incorporates not only physical but also biological characteristics of attenuation. If error from these two approaches is assumed to be similar, the differences between these two models may indicate areas where hydrologic and biologic factors are most important. For example, in headwater reaches, the v_f model predicts much higher relative attenuation compared to main stems of rivers indicating that biological factors in these small channels may be more important (Ensign and Doyle, 2006). In contrast, in lower gradient, large channels of the main stem reaches, hydrologic (physical) factors may play a larger role on instream attenuation of NO_3^- (Tank and others, 2008). However, comparing these two models is not without limitations because the assumptions and error associated with each model are likely different. This comparison does begin to indicate where physical and biological factors may be important, however, and warrants further investigation.

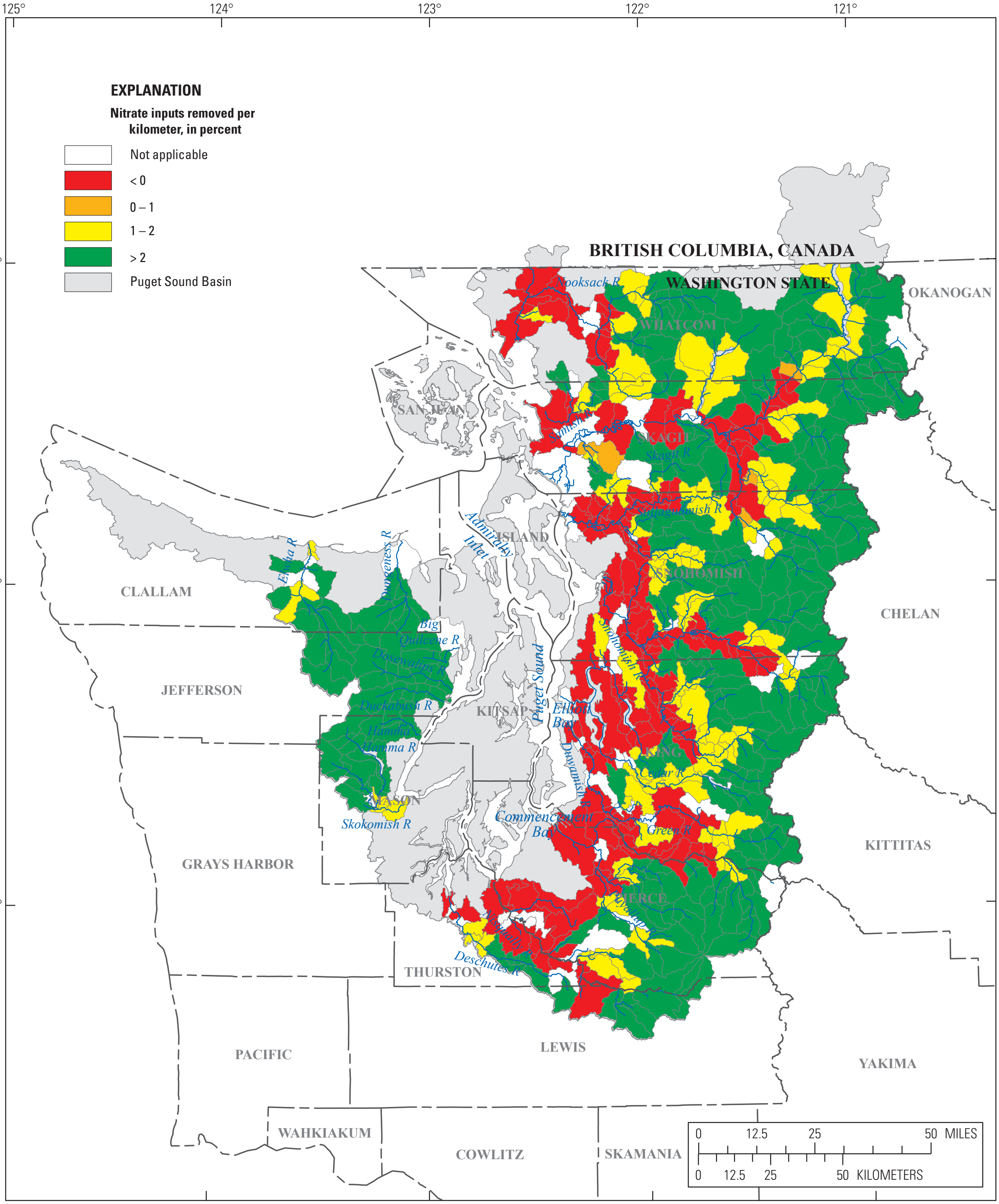
Development of a Framework for Ranking Stream Reach Attenuation

Although the RivR-N and v_f models worked well for identifying locations in the Puget Sound Basin, where relative attenuation is high or low, specific data were lacking for some of the parameters. For example, no site-specific values of v_f were available for Puget Sound stream reaches and generalized empirical relations (from other studies) were used to estimate v_f. Estimating other hydraulic parameters at the segment scale currently is the only way to calculate H_L for all reaches. Because of these limitations, a stream and river attenuation rating tool, or scorecard, was developed based on four primary factors of attenuation that can be used to quickly assess the potential for NO_3^- or PO_4^{-3} attenuation in a given reach.

Generally, nutrient removal in aquatic systems is a function of (1) the amount of biogeochemical activity, (2) the proportion of nutrient mass exposed to biogeochemically active surfaces, and (3) the duration of exposure between nutrients and these active surfaces. These three conditions can be represented by measurable parameters in stream and river reaches. Firstly, the value of v_f is a function of the strength of biogeochemical activity of a reach, it is minimally influenced by hydrology, and a robust metric to compare uptake efficiency across sites (Davis and Minshall, 1999). Secondly, the proportion of flow, and therefore proportion of nutrient load, in contact with the streambed is related to specific discharge (Q/w) or the width to depth ratio (w:d) of the channel and is shown to be related to NO_3^- retention (Lautz and Siegel, 2007; Hall and others, 2009; Hensley and others, 2014; Sheibley and others, 2014). Lastly, the duration of exposure of flow to biologically active surfaces can be related to travel time and degree of transient storage in the reach, which is related to features such as reach slope and sinuosity (Gooseff and others, 2007; Boano and others, 2014). Therefore, four primary factors of nutrient attenuation are proposed: v_f, specific discharge (Q/w), channel slope, and channel sinuosity. These factors can be used to score Puget Sound reaches into five categories to illustrate the combined potential effect of these factors on instream nutrient attenuation. Based on professional judgment and data summarized from the literature, each of these factors was given a value (a breakpoint) to split the data into two bins, one for low attenuation potential and one for high attenuation potential. Using a score of 0 for the low bin and 1 for the high bin for each of these four factors, each reach in the hydrologic network was given a cumulative score that would range from 0 (least potential for attenuation) to 4 (highest potential for attenuation).

To rank reaches based on the amount of biological activity, v_f data from a recent study that complied data from hundreds of nutrient spiraling studies for first through fifth order streams (Ensign and Doyle, 2006) were used as a guide. Small streams (first and second order) tend to have higher attenuation than large streams (fourth and fifth order) and are associated with median values of v_f for NO_3^- and PO_4^{-3} that are typically greater than 1.0 and 2.0 mm/min, respectively (Ensign and Doyle, 2006). Therefore, the breakpoints for v_f were assigned these values (table 4).

Discharge per unit channel width (Q/w), or specific discharge, indicates the contact area between water transporting nutrients and the river bed. Rivers with high width-to-depth ratios have low streamflow per unit area and allow more contact between streamflow and the river bed. Previous studies have shown that as Q/w decreases, nutrient attenuation increases (Hall and others, 2002; Lautz and Siegel, 2007; Hall and others, 2009; Hensley and others, 2014); however, there is not a clear threshold for Q/w from these studies where attenuation transitions from low to high. Therefore, to divide the reaches into approximately equal bins, the mean value of Q/w (0.5 cubic meter per second per meter ([m^3/s]/m) was used as the breakpoint for the 535 reaches in the Puget Sound hydrologic network (table 4).



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 14. Differences of relative nitrate attenuation of reaches in the hydrologic network between the v_f and RivR-N models for nitrate, Puget Sound Basin, Washington. Positive values indicate greater attenuation from the v_f model.

Table 4. Primary factors related to nutrient attenuation and criteria corresponding to a score of 1 for a given river segment.

[**Abbreviations:** $(m^3/s)/m$, cubic meter per second per meter; m/m, meter per meter; mm/min, millimeter per minute; >, greater than; <, less than]

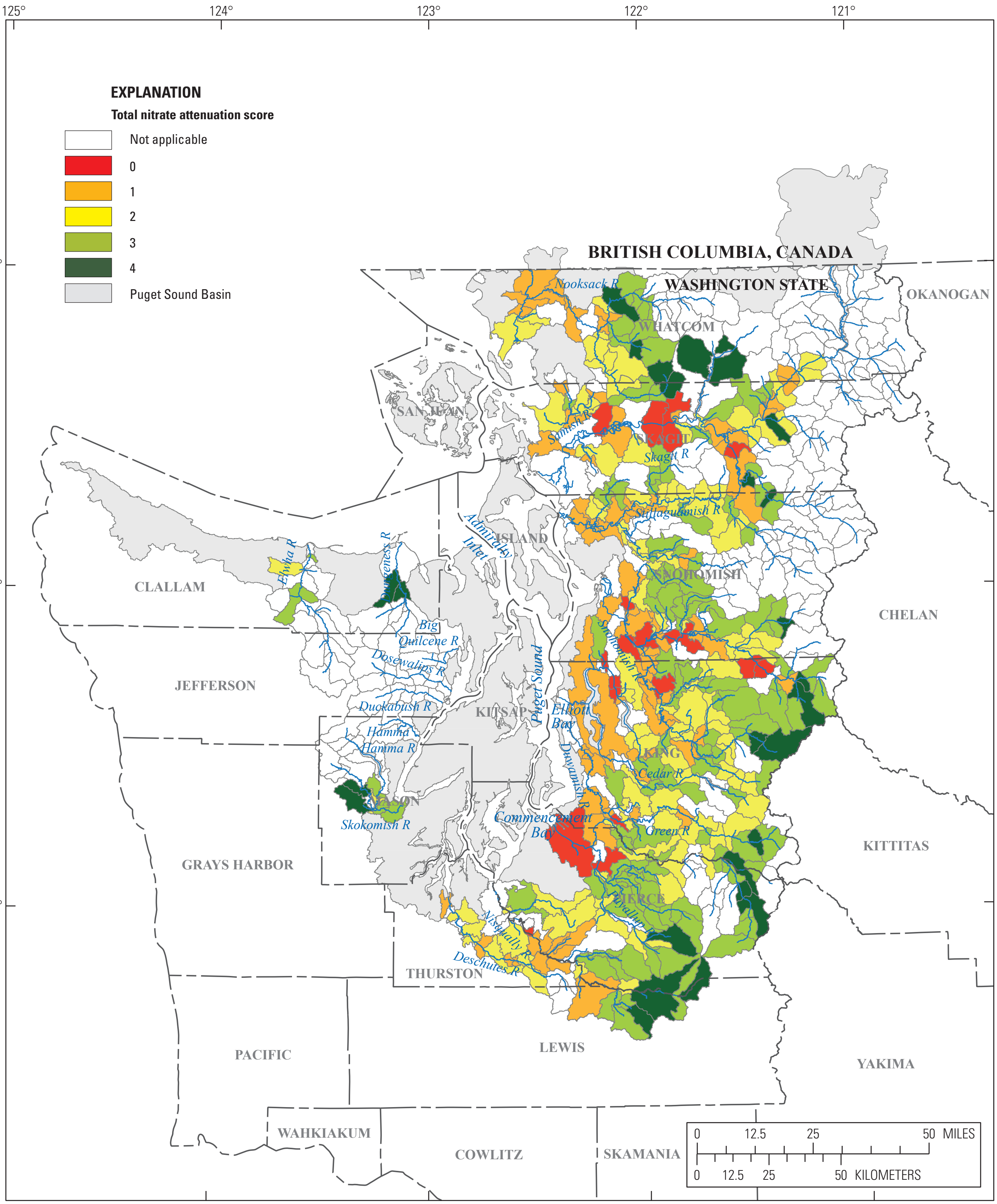
	Specific discharge [$(m^3/s)/m$]	Sinuosity (m/m)	Slope (m/m)		Uptake velocity (mm/min)	
			Nitrogen	Phosphorus	Nitrogen	Phosphorus
Criteria	< 0.5	> 1.2	> 0.005	< 0.005	> 1.0	> 2.0
Number or river segments scoring a value of 1	249	334	361	174	213	0

The water-surface gradient, or channel slope, indicates the energy available to force hyporheic flow and to transport sediment. In this case, NO_3^- attenuation is likely to be higher in high gradient rivers (all other factors equal), whereas PO_4^{-3} retention is likely to be higher in low gradient rivers through increased sedimentation of sediment-bound P. A threshold gradient of 0.005 was used to differentiate between low and high gradient river segments across the Puget Sound Basin based on studies by Beechie and others (2006); therefore, 0.005 was used as the breakpoint for scoring the reaches (table 4).

Sinuosity, which is the ratio of the path length to straight line length of river segment, indicates longer travel times through a segment, pool-riffle morphology, and presence of point bars. Hyporheic flow is associated with pool-riffle morphology and point bars and is caused by changes in localized vertical hydraulic gradients. A sinuosity of 1.2 generally is used to distinguish sinuous channels from straight channels (Knighton, 1998); therefore, 1.2 was used as the breakpoint for this factor in the analysis (table 4).

Values for v_f, NO_3^-, and PO_4^{-3} for each river segment were the same data used for the v_f model (see section vf attenuation model). The Q/w for each reach was calculated from the mean annual discharge for each reach used in the RivR-N and v_f models (derived from fig. 6) and GIS-based measurements of average channel width (w) that were based on high-resolution elevation data (Konrad, 2015). Average channel width from Konrad (2015) was used instead of estimating width using equation 5 (Magirl and Olsen, 2009) because the former represented more accurate representation of field conditions. However, the widths from Konrad (2015) were not available for approximately 200 of the 535 river segments, which limited the ability to score this factor across all reaches of the river network. Despite this limitation, it was determined that using fewer but more-accurate data was warranted and strengthened the scoring analysis. Data for channel slope (water-surface gradient) and sinuosity for the 535 reaches of the river network were derived as part of a geospatial assessment of flood plains along major rivers in the Puget Sound Basin (Konrad, 2015). Surface-water gradient was used as a proxy for channel slope and was calculated by the change in surface-water elevation extrapolated from the National Elevation Dataset (U.S. Geological Survey, 2012) from the top and bottom of each river segment (Konrad, 2015). Sinuosity was calculated by measuring the length of each river segment divided by the straight-line distance between the endpoints of the segment (Konrad, 2015).

For NO_3^- attenuation, 328 of the 535 reaches (fig. 15) could be scored, and 207 reaches lacked at least one of the four factors needed to fully score the reach. Only a handful of reaches (25 of 328) scored the highest value of four, indicating the greatest attenuation potential. These reaches were in locations similar to reaches with high calculated relative attenuation from the RivR-N and v_f models, namely, in headwater reaches of the major river basins. Reaches scoring a four indicate that biological activity likely is high (v_f for NO_3^- greater than 1.0 mm/min), have steep gradients, sinuous channels, and a high proportion of surface water in contact with streambed sediments. Most reaches scored in the middle with 213 reaches scoring either a two or three, indicating a moderate potential for NO_3^- attenuation. Only 19 reaches scored a zero and were usually located on areas of the main stems of larger rivers, which indicated that the primary factors needed for attenuation were not sufficient. This does not necessarily mean that no attenuation is occurring in these reaches, but the potential for attenuation is low.



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 15. Total nitrate attenuation scores in streams and rivers, Puget Sound Basin, Washington.

For PO_4^{-3} attenuation, 327 of the 535 reaches could be scored with 208 reaches missing data for at least one of the four factors (fig. 16). The most interesting result for the PO_4^{-3} scoring was that no reach scored a total of four because none of the reaches had a PO_4^{-3} v_f value greater than the breakpoint of 2.0 mm/min. This may indicate that reaches in the Puget Sound Basin are not efficient at biological P removal relative to data summarized in the literature. However, the accuracy of the method used to calculate v_f for PO_4^{-3} is uncertain. Of the 327 reaches, 55 reaches scored a three, and were spread throughout the major river basins (fig. 16). There tended to be more high scores for PO_4^{-3} compared to NO_3^- in main stem reaches, likely a result of lower slopes and increased potential for sedimentation. A total of 261 of the 328 reaches scored a one or two, which indicates that this scoring procedure ranks many reaches in the Puget Sound Basin low for potential PO_4^{-3} attenuation. This does not necessarily mean PO_4^{-3} potential in Puget Sound rivers is not important, but might indicate that a different set of factors would describe potential PO_4^{-3} attenuation better if there were limitations in scoring v_f for PO_4^{-3} from the inconsistent variance structure (heteroscedasticity) in the regression equation used to calculate PO_4^{-3} from TP (fig. 9*B*). The distribution of scores in the Puget Sound Basin for each attenuation factor individually is provided in appendix B.

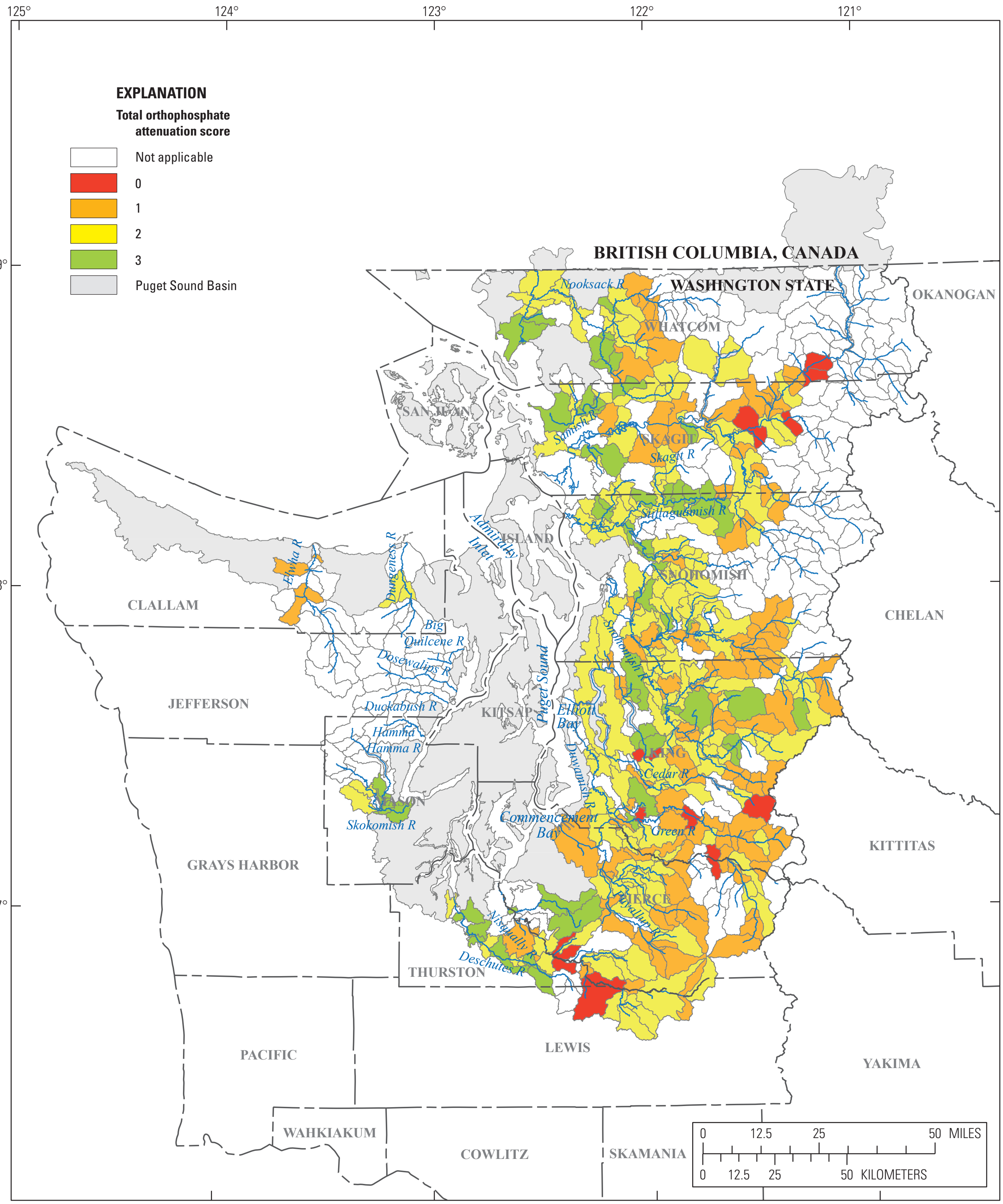
Many similarities were noted in the patterns of NO_3^- and PO_4^{-3} attenuation when comparing the reach scores (figs. 15 and 16) to the results from the RivR-N and two v_f models (figs. 7, 10, and 12). The relative attenuation values calculated using the RivR-N and v_f models were compared to the scores generated from the ranking approach. The comparison indicated that the scoring procedure accurately identified reaches with high and low attenuation potential (fig. 17). For the RivR-N and NO_3^- v_f models, median relative attenuation increased as reach scores increased and this relation is stronger for the v_f model (fig. 17*B*) than the RivR-N model (fig. 17*A*). However, for PO_4^{-3}, the relation between the relative attenuation and reach score was not as obvious (fig. 17*C*), indicating that the scoring procedure is not as effective for PO_4^{-3}, or that other factors are more important for describing PO_4^{-3} attenuation that were not included in the ranking.

Temporal Effects of Nutrient Attenuation

The analysis of relative nutrient attenuation was focused on using average annual discharges and nutrient concentrations for the hydrologic network. However, hydrologic conditions and nutrient concentrations can change dramatically during the year which, in turn, will influence the amount of nutrient attenuation. Therefore, to evaluate the temporal variability of changes in flow and concentration on relative nutrient attenuation, the v_f model was examined in more detail for three case studies using sites with long-term USGS water-quality data in the Puget Sound Basin. An intermediate sized, high-gradient reference stream (North Fork Skokomish), a small sized, low-gradient urban stream (Thornton Creek), and large sized, low-gradient river in the north Puget lowlands (Nooksack River) were selected. These three sites were selected to span a range of both hydrologic (discharge, channel width) and nutrient (concentration) characteristics.

The North Fork Skokomish River site is in the Olympic National Park. It is a high-gradient, forested site with a range of substrate sizes, but generally is dominated by cobble and small boulders. The North Fork Skokomish River drains an area of 57.2 mi^2 and is located entirely within the Olympic National Park. The high-flow period is typically from October through January. The transition period, from February through July, is a time of generally lower streamflows than in winter and diminishing flows during summer. The low-flow period is from August through September. Mean daily discharge, based on 89 years of record, are 673 ft^3/s with a maximum of 9,980 ft^3/s in 1956 and a minimum of 37 ft^3/s in 1937. Based on previous studies, the highest concentrations of TN, TP, and suspended sediment tended to occur during the high-flow period (Embrey and Frans, 2003). Median NO_3^- concentrations from 1996 to 2014 were 0.05 µg/L and ranged from less than the detection limit of 0.05 µg/L to 0.17 µg/L. Orthophosphate concentrations from 1996 to 2014 often were less than the detection limit of 0.01 µg/L and ranged to 0.06 µg/L with a median of <0.01 µg/L (Ebbert and others, 2000; Embrey and Frans, 2003).

The Thornton Creek site is in a subbasin of the Lake Washington drainage and entirely within the Puget Sound lowlands physiographic province. Population density in the basin is high, on the order of 600–1,000 people per square mile, an example of a fully developed urban basin. It is largely residential, but includes large shopping malls and commercial development and an interstate freeway that bisects the basin. The creek drains an area of 12.1 mi^2. Its substrate is dominated by silts and sands with some gravels. Mean annual discharge is about 9.6 ft^3/s from 1996 to 2013. Low flows occur in the summer and high flows occur during late autumn through spring, with periodic storm-related high flows. Historical water-quality sampling of this site has detected elevated concentrations of numerous chemicals. Nitrate concentration ranged from 0.42 to 1.70 mg/L with a median of 1.3 mg/L from 1996 to 2011. During this same period, the median concentration of PO_4^{-3} was 0.03 mg/L and ranged from less than the detection limit of 0.01 mg/L to 0.3 mg/L (Ebbert and others, 2000; Embrey and Frans, 2003).



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure 16. Total orthophosphate attenuation scores in streams and rivers, Puget Sound Basin, Washington.

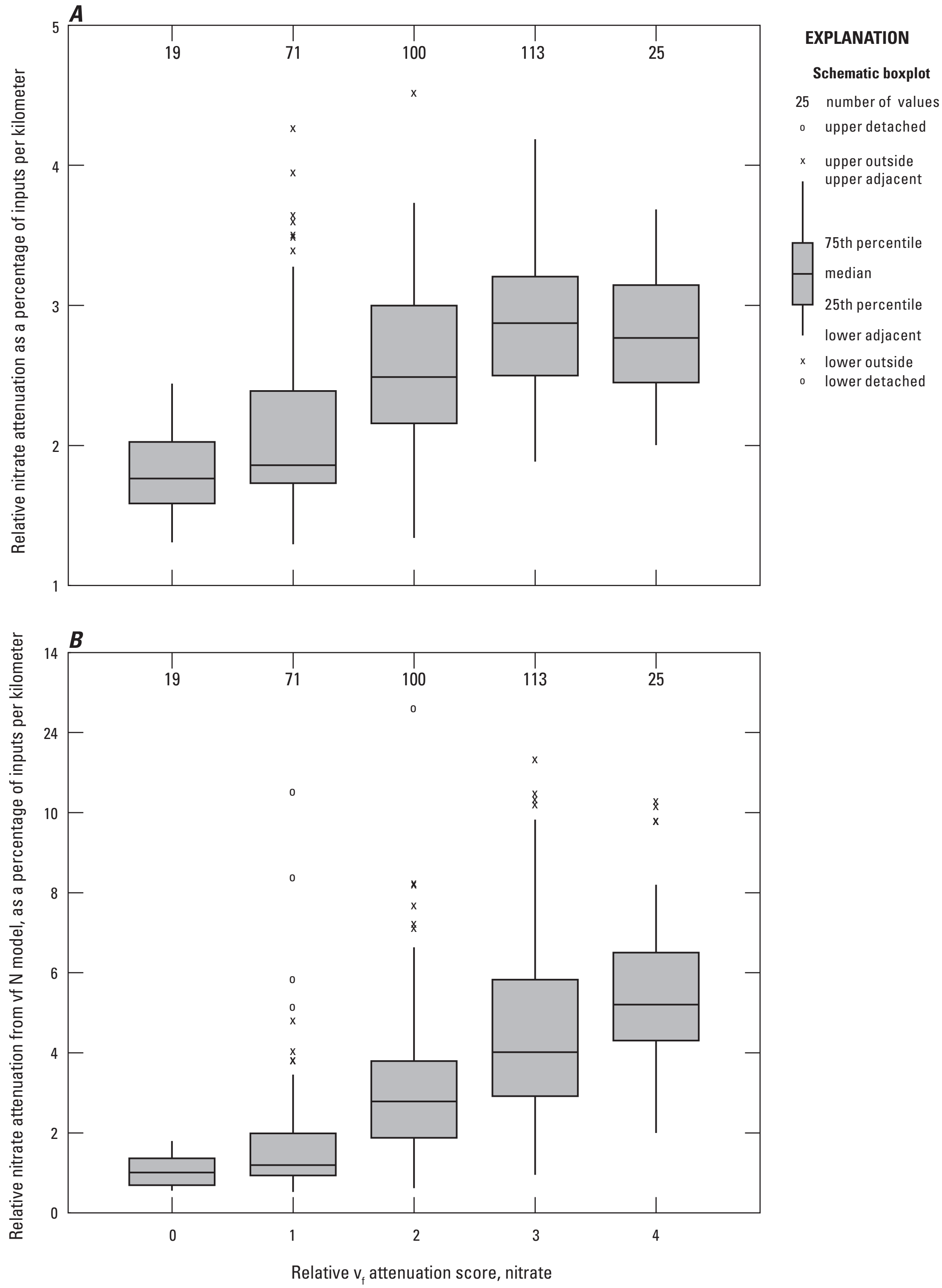


Figure 17. Relation between relative nutrient attenuation and reach scores for the (*A*) RivR-N model, (*B*) v_f nitrogen model, and (*C*) v_f orthophosphate model, Puget Sound Basin, Washington.

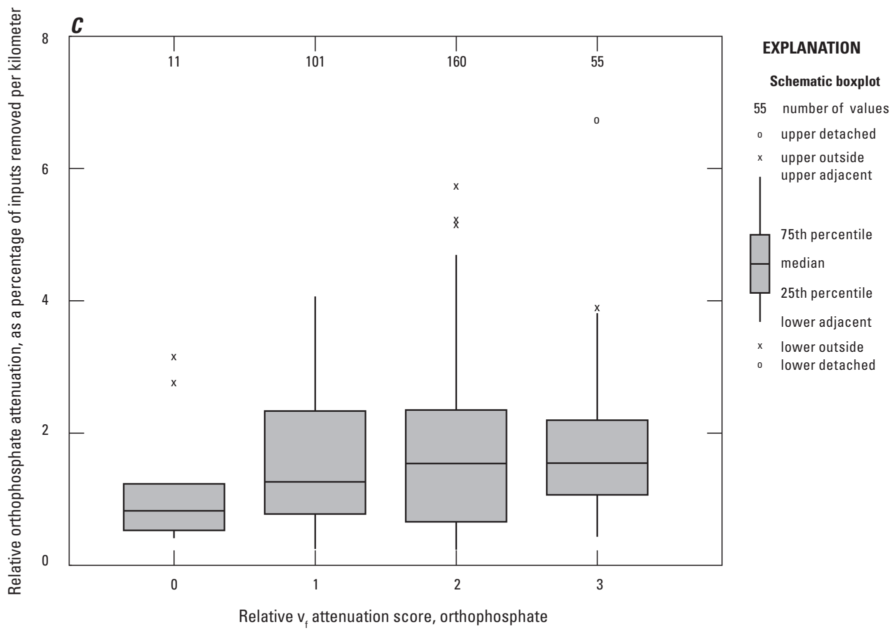


Figure 17.—Continued

The Nooksack River site at Ferndale drains an area of about 790 mi^2 of a watershed dominated by forest practices in the headwaters (about 70 percent of the watershed) and agriculture and urbanization in the downstream portion of the watershed near the site. The river substrate is dominated by fines and gravels. Average daily discharges at the site based on USGS streamgage data (12213100) are 4,140 ft^3/s, but have exceeded 20,000 ft^3/s and have been as low as 593 ft^3/s during the period of record (1986–2014). The dry season is generally during late summer; the highest flows typically occur in the spring, but can increase rapidly during storm events. Low-nutrient water from the mountain headwaters has an important influence on the water chemistry of the main stem Nooksack River and contributes to the dilution of high nutrient inputs to this river from urban and agricultural land uses in the watershed (fig. 2). At times, however, large volumes of runoff can result in high flows with large amounts of sediment and occasionally high concentrations of TP.

Historical samples collected at this site (from 1995 to 1998) have NO_3^- concentrations ranging from 0.13 mg/L to a maximum of 0.92 mg/L, with a median of 0.35 mg/L (Ebbert and others, 2000; Embrey and Frans, 2003). Orthophosphate concentration ranged from less than the detection limit of 0.01 mg/L to a maximum of 0.06 mg/L and a median of <0.01 mg/L during the same period. A summary of nutrient concentrations at these sites compared to the range reported for Puget Sound is presented in table 5.

Measured hydrologic parameters were downloaded from NWIS for a representative low- and high-flow condition at each site since 2000 (table 6) and used to calculate H_L (Q/wL). Discharge, stream width, average depth, and cross sectional area are measured and reported whenever a manual discharge measurement is made and the average velocity was calculated by taking discharge divided by the cross sectional area.

Table 5. Nutrient data for three study sites compared to nutrient data for Puget Sound Basin, Washington.

[Concentrations are in milligram per liter. –, not calculated; <, less than]

Site name	Site No.	Nitrate			Orthophosphate		
		Minimum	Maximum	Median	Minimum	Maximum	Median
North Fork Skokomish River	12056500	<0.05	0.17	0.05	<0.01	0.06	<0.01
Thornton Creek	12128000	0.42	1.7	1.3	<0.01	0.3	0.03
Nooksack River at Ferndale	12213100	0.13	0.92	0.35	<0.01	0.06	0.01
Puget Sound Basin[1]		<0.05	7	–	<0.01	1	–

[1] From data summarized in Ebbert and others (2000) and Embrey and Frans (2003).

Table 6. Representative high- and low-flow conditions at the three case study sites to estimate nutrient attenuation, Puget Sound Basin, Washington.

[**Abbreviations:** ft^3/s, cubic foot per second; ft^2, foot squared;, ft, foot; ft/s, foot per second]

Site name	Date	Flow category	Discharge (ft^3/s)	Cross sectional area (ft^2)	Width (ft)	Depth (ft)	Uptake rate (ft/s)
North Fork Skokomish River	08-28-13	Low	140	88	136	0.6	0.9
	11-14-01	High	2,450	466	107	4.4	4.6
Thornton Creek	07-19-11	Low	4	3	7	0.5	1.3
	11-15-01	High	28	17	14	1.2	1.7
Nooksack River at Ferndale	10-29-13	Low	1,140	1,020	198	5.2	1.1
	01-14-14	High	10,900	2,810	236	11.9	3.9

A standardized reach length (L) of 1,000 m was used to calculate H_L in order to compute relative attenuation across these sites because measurements were made at a single locations within a much longer (and undefined) reach. Therefore, this analysis is consistent with the approach used previously when using the RivR-N and v_f models. Values of v_f for use in equation 6 were based on the concentration relations in equations 8 and 9 using the range of nutrient concentrations at each site and for the range in the Puget Sound Basin (table 5). For NO_3^- and PO_4^{-3}, relative attenuation was always higher during low-flow conditions compared to high-flow conditions even when concentrations were high (figs. 18 and 19). For NO_3^-, the slope of the relative attenuation curves approach zero at about 3 mg N/L indicating that at concentrations greater than 3 mg N/L, uptake in the reaches may be saturated and relative attenuation approaches a minimum (fig. 18). During high flow, relative NO_3^- attenuation is minimal and the saturation point of attenuation is about 1 mg N/L (fig. 18), much lower than during low-flow conditions. For PO_4^{-3}, saturation of relative attenuation for both high- and low-flow conditions was similar and starts at about 0.1 mg P/L (fig. 19).

When relative retention is plotted for only the parts of each curve represented by the measured range in nutrient concentrations at each site, some interesting features are observed (figs. 20 and 21). The North Fork Skokomish relative NO_3^- attenuation begins high and decreases rapidly as concentration increases for both the low and high flow conditions (fig. 20). The Nooksack relative NO_3^- retention is fairly constant over the range of observed concentrations and always low for the low-and high-flow conditions. Thornton Creek shows small but not dramatic changes in relative retention over the range of observed concentrations. During low flow, relative NO_3^- attenuation in Thornton Creek is in the middle of the other two sites, but shows the highest relative attenuation during high flow, probably because the difference in hydrologic conditions between low and high-flow conditions at this site are smaller than the other two sites (table 6). For relative PO_4^{-3} attenuation, North Fork Skokomish River and Thornton Creek were similar during low flow, but during high flow, relative attenuation was greater at Thornton Creek and the North Fork Skokomish River was negligible and similar to the Nooksack River. Relative PO_4^{-3} attenuation in the Nooksack River was always small and never exceeded 1 percent of inputs attenuated per kilometer (fig. 21).

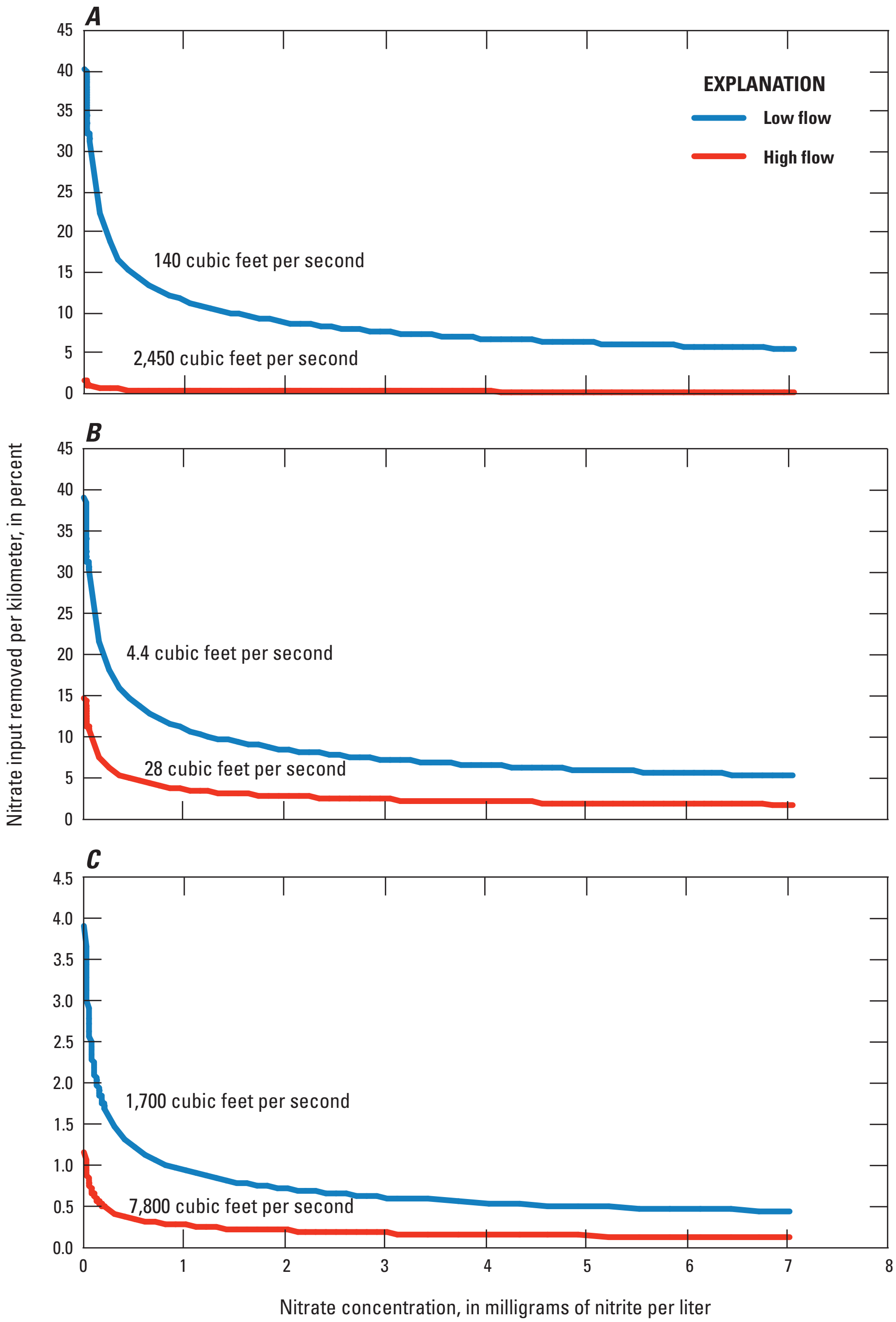


Figure 18. Relation between relative nitrate attenuation and nitrate concentration for representative high- and low-flow conditions at (*A*) North Fork Skokomish River, (*B*) Thornton Creek, and (*C*) Nooksack River at Ferndale, Washington. The nitrate concentration range on the x-axis represents the overall range detected in the Puget Sound Basin (Ebbert and others, 2000; Embrey and Frans, 2003).

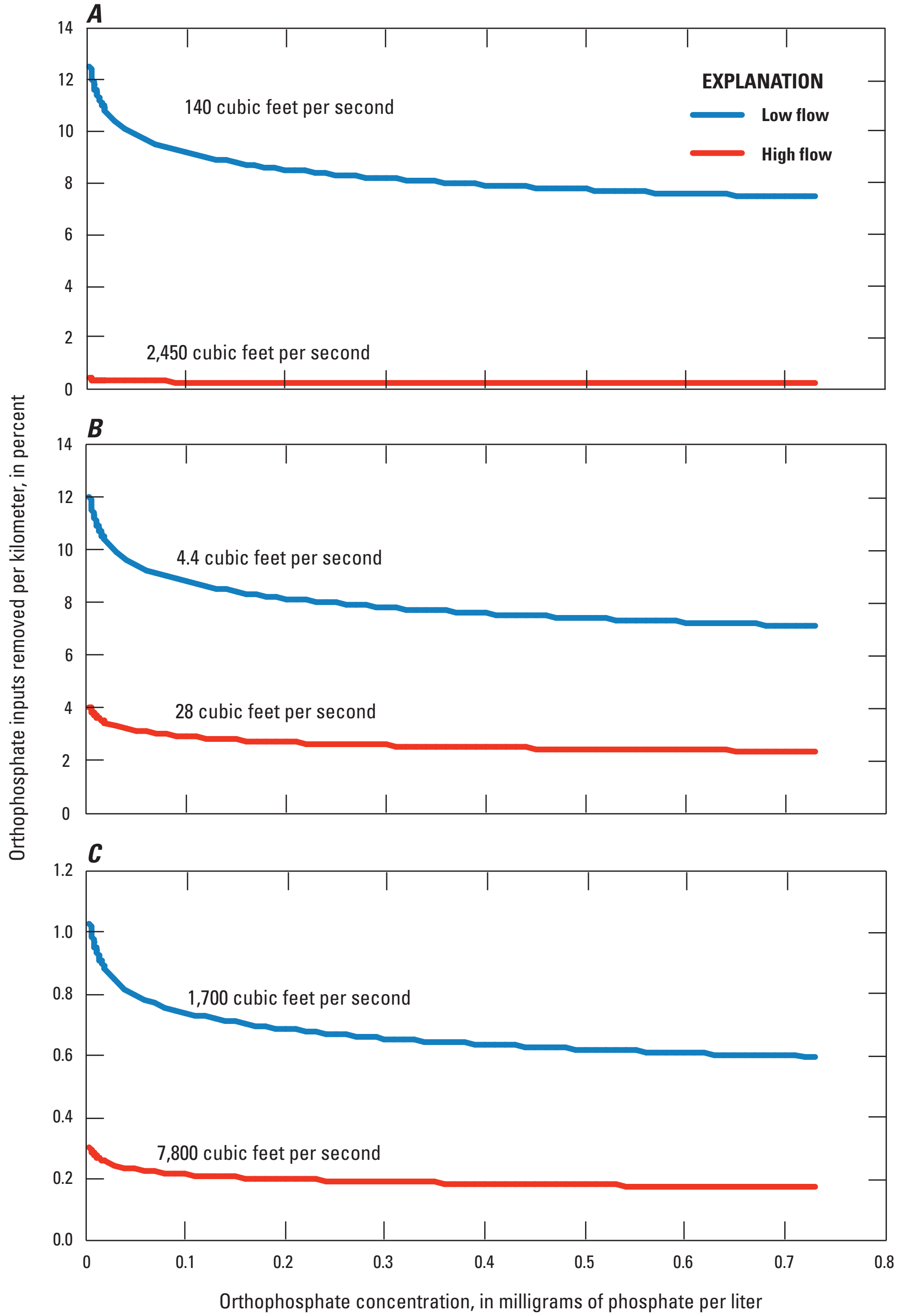


Figure 19. Relation between relative orthophosphate attenuation and orthophosphate concentration for representative high- and low-flow conditions at (*A*) North Fork Skokomish, (*B*) Thornton Creek, and (*C*) Nooksack River at Ferndale, Washington. The orthophosphate concentration range on the x-axis represents the overall range detected in the Puget Sound Basin (Ebbert and others, 2000; Embrey and Frans, 2003).

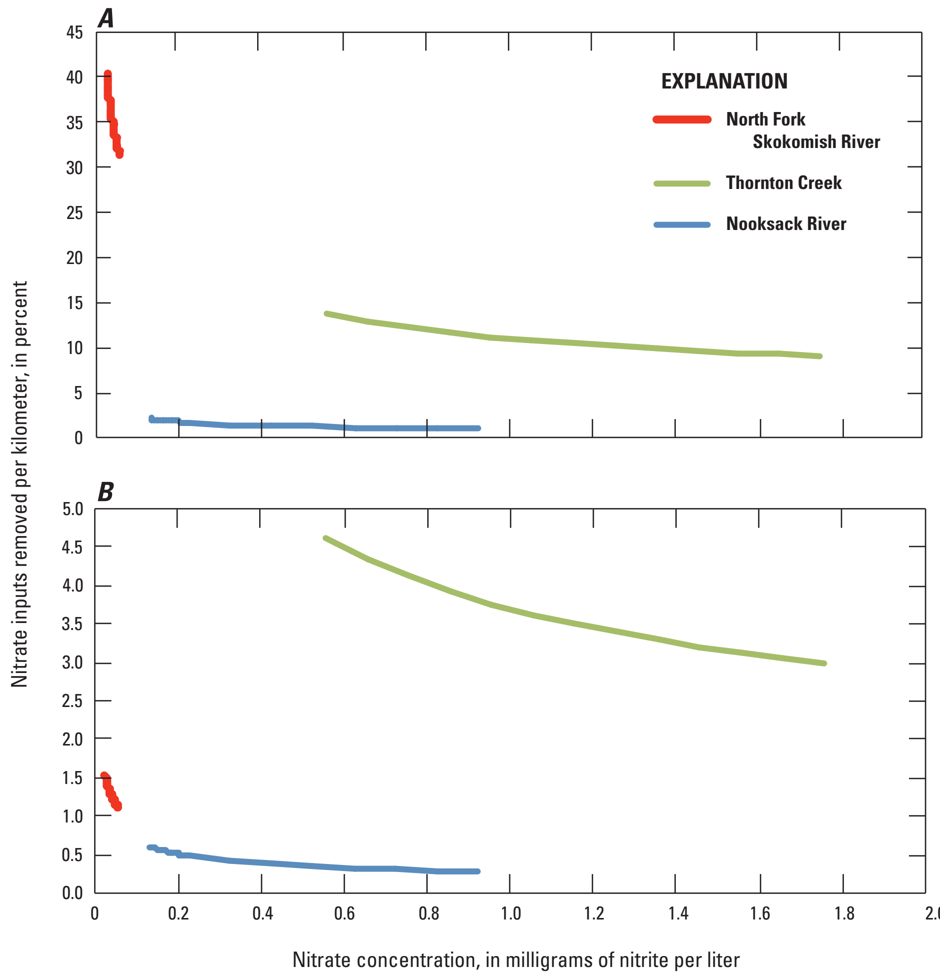


Figure 20. Relative nitrate attenuation and nitrate concentration for the range in nutrient concentrations for typical (*A*) low- and (*B*) high-flow conditions, North Fork Skokomish River, Thornton Creek, and Nooksack River near Ferndale, Washington.

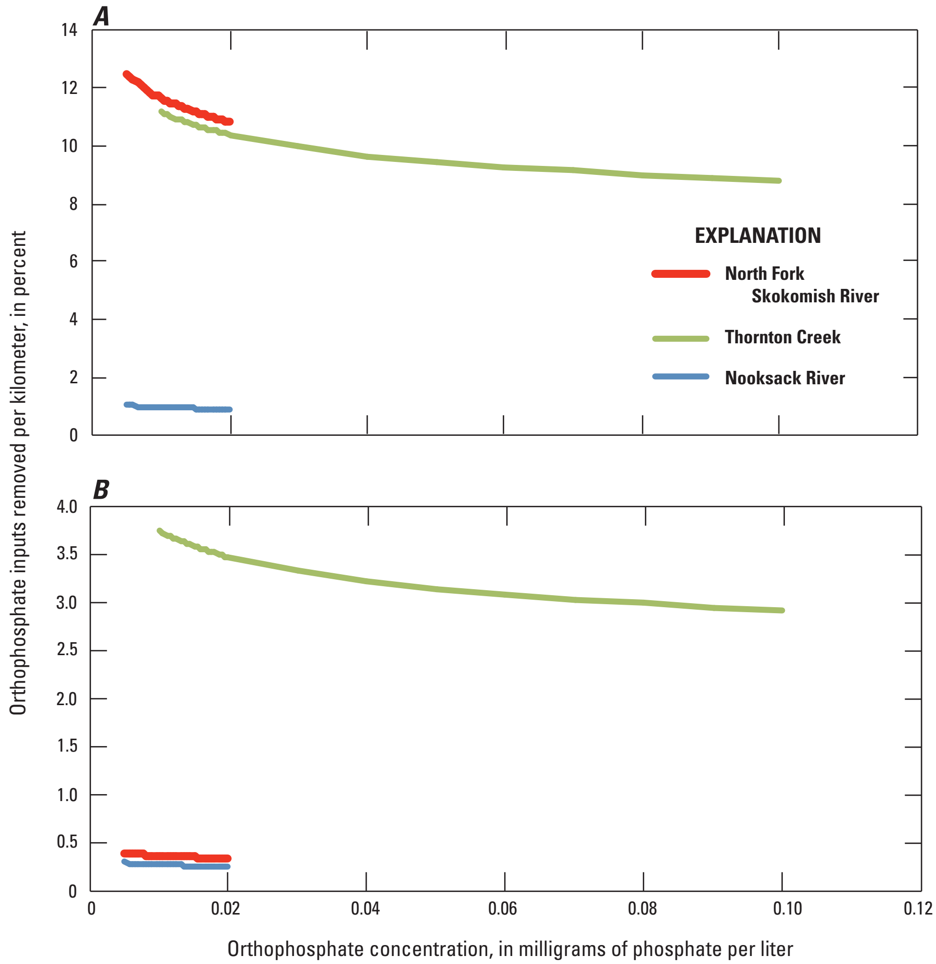


Figure 21. Relative orthophosphate attenuation and orthophosphate concentration for the range in nutrient concentrations for typical (*A*) low- and (*B*) high-flow conditions, North Fork Skokomish River, Thornton Creek, and Nooksack River near Ferndale, Washington.

Seasonality of relative attenuation at these three sites was determined from field measured data downloaded from NWIS for discharge, stream width, and concentrations of NO_3^- and PO_4^{-3} (tables 7 and 8), and summarized monthly to identify when attenuation is greatest during the course of a year. Because attenuation calculated by the v_f model (equation 6) varies with nutrient concentration and H_L (Q/wL), monthly variability of relative attenuation was examined by (1) holding concentration constant at the monthly median value and using a the monthly range of hydrologic conditions and (2) holding hydrologic conditions constant at the monthly median value and using the monthly range of concentrations. This approach was used to examine the variability in monthly relative attenuation from all parameters. However, the variability of relative attenuation from hydrologic parameters was always greater than the variability from changes in monthly nutrient concentration. Therefore, only seasonal relative attenuation data based on the monthly range of hydrologic conditions at each site are presented.

Table 7. Discharge and stream width measures by month for the North Fork Skokomish River, Thornton Creek, and Nooksack River, Washington.

[North Fork Skokomish, 178 observations from 1986 to 2014; Thornton Creek, 158 observations from 1996 to 2013; and Nooksack River, 177 observations from 1987 to 2014. **Abbreviations:** ft, foot; ft³/s, cubic foot per second]

Site name	Month	Discharge (ft³/s)				Width (ft)			
		Minimum	Maximum	Median	Mean	Minimum	Maximum	Median	Mean
North Fork Skokomish River	Jan.	114	9,660	596	913	65	168	108	114
	Feb.	133	3,500	446	576	64	179	105	103
	Mar.	165	7,300	414	606	76	136	110	109
	Apr.	172	3,940	461	543	56	165	118	115
	May	231	2,760	585	665	56	140	125	107
	June	169	2,380	502	574	62	176	107	109
	July	84	2,460	253	341	78	128	100	102
	Aug.	48	1,840	120	165	70	136	87	94
	Sept.	40	2,870	82	139	70	112	78	83
	Oct.	38	7,620	194	420	56	129	83	92
	Nov.	39	5,440	555	900	81	140	111	111
	Dec.	119	13,100	510	840	59	172	106	113
Thornton Creek	Jan.	5.1	81.0	14.0	15.8	6.5	16.1	13.7	12.2
	Feb.	4.4	42.0	10.0	12.1	7.0	17.4	12.7	12.3
	Mar.	4.6	68.0	10.0	12.7	6.3	16.2	10.0	10.9
	Apr.	4.3	55.0	8.3	10.2	6.9	17.5	10.0	11.0
	May	3.5	45.0	6.8	8.2	5.9	16.4	12.0	11.4
	June	0.8	38.0	5.1	6.2	6.4	18.0	7.9	10.5
	July	2.2	24.0	3.9	4.6	5.2	15.3	9.4	9.9
	Aug.	0.6	33.0	3.5	4.1	6.1	17.4	10.6	11.2
	Sept.	0.4	39.0	4.2	5.5	7.4	16.7	11.7	11.6
	Oct.	2.8	116.0	5.8	8.6	6.3	21.0	11.9	12.0
	Nov.	3.8	104.0	9.5	13.5	7.0	21.5	14.3	13.7
	Dec.	3.1	190.0	12.0	15.1	8.0	24.0	13.3	14.2
Nooksack River	Jan.	1,570	47,500	4,230	5,557	203	395	236	256
	Feb.	1,180	27,500	3,090	3,763	199	223	212	211
	Mar.	1,170	33,700	3,195	4,123	206	256	218	223
	Apr.	1,720	15,700	3,515	4,001	193	236	215	214
	May	2,200	15,300	4,540	4,991	212	230	218	220
	June	2,120	17,700	4,675	4,944	194	300	214	217
	July	1,650	15,300	3,035	3,538	200	226	213	214
	Aug.	852	10,700	1,810	2,080	190	308	205	208
	Sept.	672	13,800	1,310	1,769	197	353	211	242
	Oct.	565	32,300	2,030	3,080	180	293	212	215
	Nov.	575	42,100	4,195	5,570	210	397	239	262
	Dec.	1,200	32,300	3,790	4,830	200	279	217	220

Table 8. Nitrate and orthophosphorus data by month for the North Fork Skokomish River, Thornton Creek, and Nooksack River at Ferndale, Washington.

[North Fork Skokomish, 87 observations from 1996 to 2014; Thornton Creek, 164 observations from 1996 to 2011; and Nooksack River, 41 observations from 1995 to 1998. **Abbreviatons:** mg/L, milligram per liter]

Site name	Month	Nitrate (mg/L)				Orthophosphate (mg/L)			
		Minimum	Maximum	Median	Mean	Minimum	Maximum	Median	Mean
North Fork Skokomish River	Jan.	0.024	0.090	0.030	0.040	0.009	0.061	0.022	0.026
	Feb.	0.025	0.147	0.034	0.068	0.006	0.034	0.016	0.020
	Mar.	0.025	0.030	0.025	0.027	0.006	0.031	0.016	0.019
	Apr.	0.021	0.100	0.030	0.046	0.006	0.028	0.016	0.014
	May	0.025	0.058	0.030	0.035	0.006	0.055	0.016	0.020
	June	0.005	0.087	0.030	0.033	0.006	0.031	0.016	0.020
	July	0.023	0.070	0.030	0.033	0.009	0.037	0.023	0.021
	Aug.	0.019	0.054	0.030	0.034	0.006	0.031	0.011	0.013
	Sept.	0.025	0.090	0.032	0.042	0.009	0.031	0.022	0.021
	Oct.	0.025	0.090	0.080	0.062	0.006	0.016	0.016	0.012
	Nov.	0.025	0.168	0.050	0.056	0.009	0.061	0.016	0.024
	Dec.	0.020	0.043	0.025	0.027	0.006	0.031	0.017	0.020
Thornton Creek	Jan.	0.477	1.720	1.320	1.299	0.049	0.144	0.071	0.079
	Feb.	0.680	1.660	1.350	1.254	0.061	0.129	0.071	0.078
	Mar.	0.417	1.590	1.095	1.021	0.031	0.184	0.067	0.076
	Apr.	0.480	1.300	0.969	0.940	0.031	0.156	0.061	0.071
	May	0.553	1.400	1.110	1.069	0.016	0.304	0.086	0.096
	June	0.656	1.480	1.095	1.054	0.040	0.166	0.097	0.105
	July	0.727	1.380	1.070	1.059	0.046	0.193	0.149	0.144
	Aug.	0.816	1.300	0.961	1.010	0.046	0.190	0.147	0.132
	Sept.	0.570	1.410	0.928	0.975	0.069	0.190	0.137	0.129
	Oct.	0.630	1.400	1.130	1.073	0.086	0.236	0.123	0.129
	Nov.	0.620	1.450	1.005	1.010	0.046	0.156	0.094	0.098
	Dec.	0.430	1.580	1.220	1.167	0.055	0.153	0.079	0.085
Nooksack River	Jan.	0.350	0.841	0.558	0.577	0.016	0.061	0.040	0.039
	Feb.	0.510	0.920	0.573	0.668	0.016	0.055	0.031	0.034
	Mar.	0.230	0.750	0.712	0.601	0.016	0.049	0.023	0.028
	Apr.	0.270	0.540	0.469	0.437	0.016	0.031	0.016	0.019
	May	0.139	0.450	0.278	0.286	0.016	0.016	0.016	0.016
	June	0.183	0.250	0.217	0.217	0.016	0.031	0.023	0.023
	July	0.142	0.229	0.170	0.178	0.016	0.031	0.023	0.023
	Aug.	0.128	0.200	0.164	0.164	0.016	0.061	0.038	0.038
	Sept.	0.230	0.370	0.275	0.292	0.016	0.061	0.016	0.031
	Oct.	0.207	0.375	0.210	0.264	0.016	0.016	0.016	0.016
	Nov.	0.257	0.560	0.260	0.369	0.016	0.034	0.031	0.029
	Dec.	0.570	0.920	0.604	0.698	0.016	0.061	0.016	0.031

For both NO_3^- and PO_4^{-3}, relative attenuation was greatest in July through September and was lowest and relatively constant from January through June (figs. 22 and 23). Variability in monthly relative attenuation is also the greatest in July through September. Peaks in relative nutrient attenuation at all three sites correspond to times when monthly mean flows are the lowest and monthly mean nutrient concentration declines (tables 7 and 8). The combination of lower flows and nutrient concentrations leads to longer travel times within the reach, more contact time of surface water with the streambed (from lower Q/w), and more efficient relative attenuation (via lower concentration and higher v_f). The summer peak in relative nutrient attenuation is greatest for the North Fork Skokomish site likely because the change to lower flows are the most pronounced for this site.

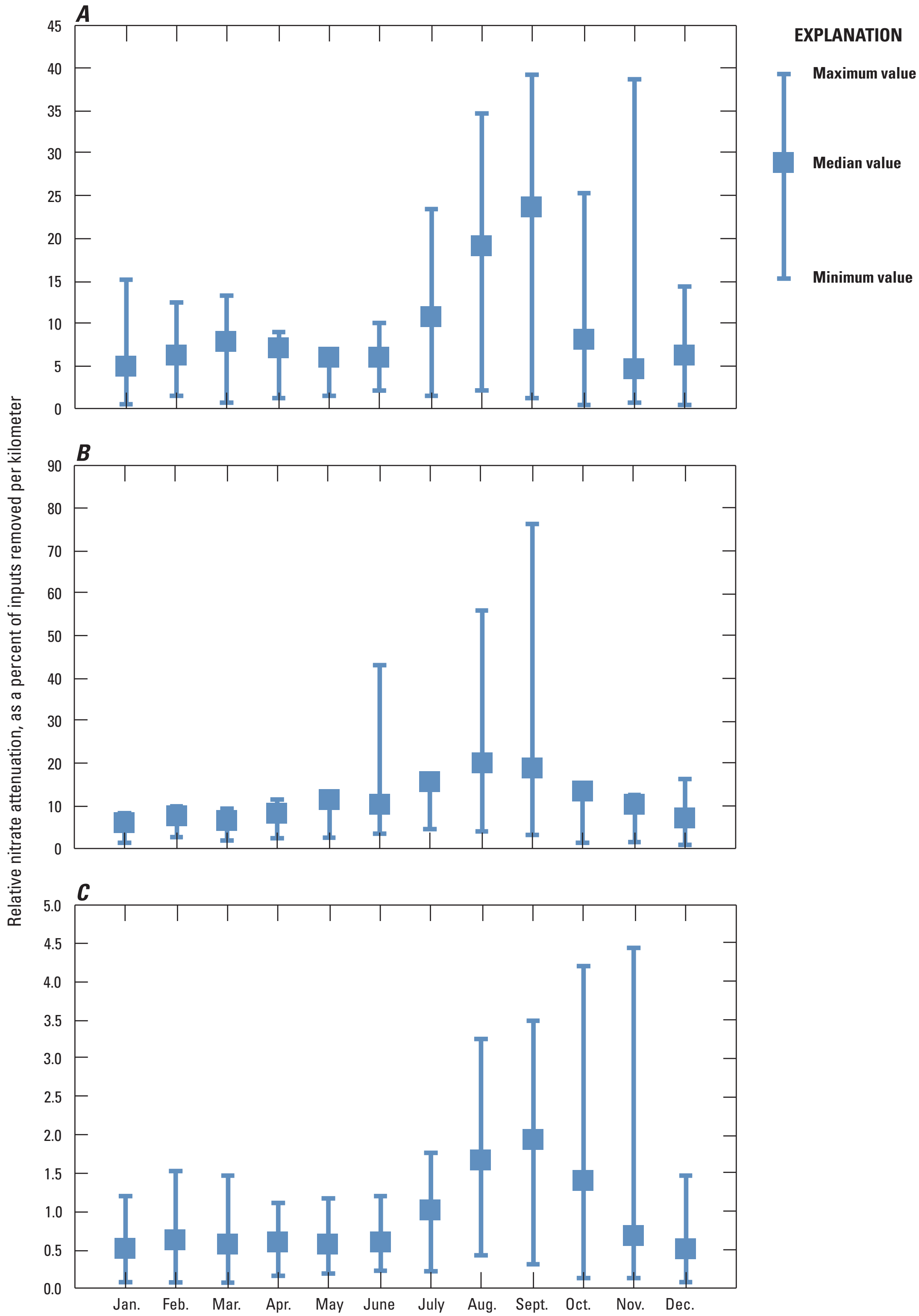


Figure 22. Monthly variation of relative nitrate attenuation using the v_f model given by equation 6 for the (*A*) North Fork Skokomish River, (*B*) Thornton Creek, and (*C*) Nooksack River at Ferndale, Washington. Data points represent median monthly relative attenuation and bars represent the minimum and maximum values based on variation in monthly stream discharge and width.

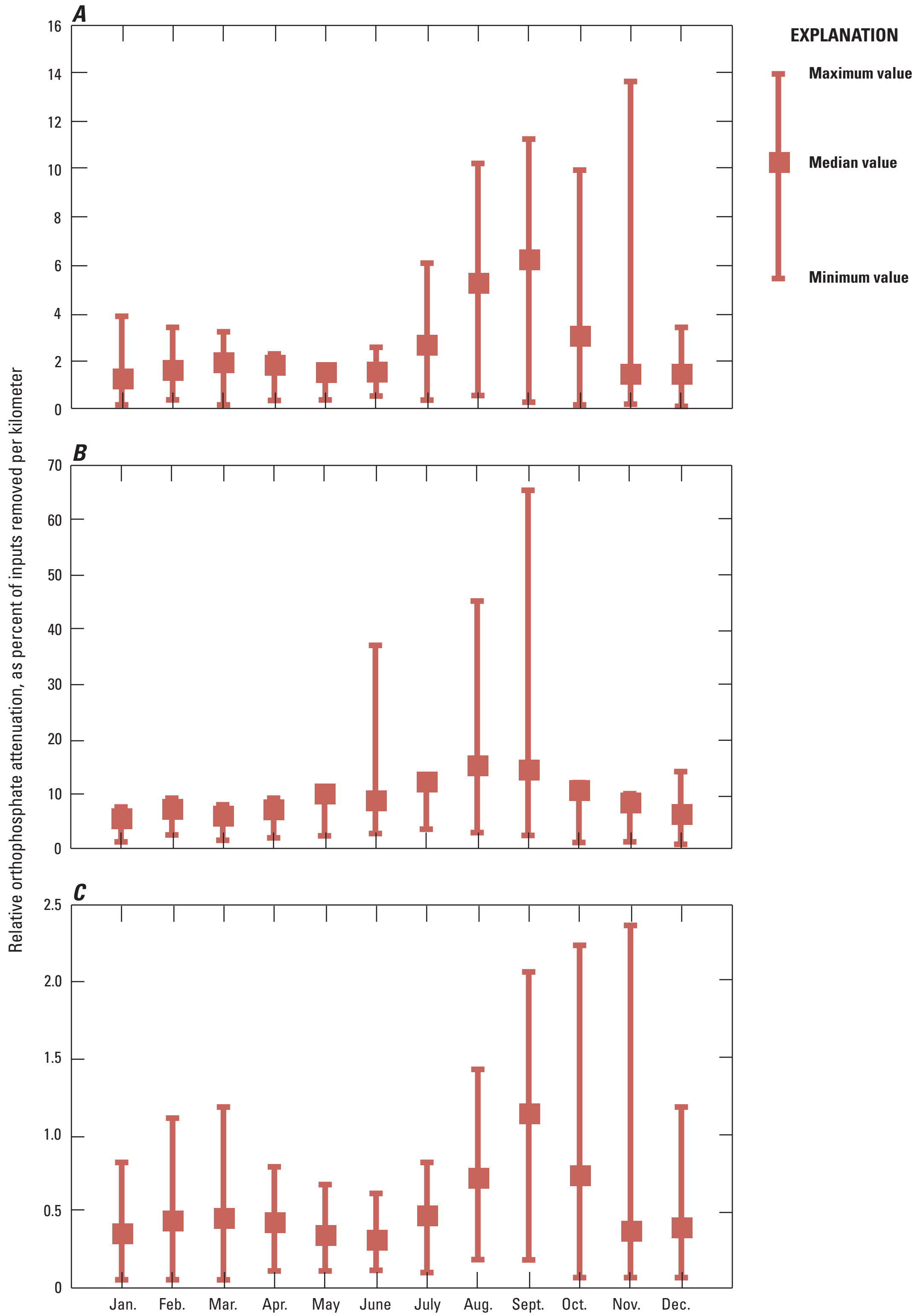


Figure 23. Monthly variation of relative orthophosphate attenuation using the v_f model given by equation 6 for the (*A*) North Fork Skokomish River, (*B*) Thornton Creek, and (*C*) Nooksack River at Ferndale, Washington. Data points represent median monthly relative attenuation and bars represent the minimum and maximum values based on variation in monthly stream discharge and width.

Across all months, relative attenuation is about an order of magnitude lower in the Nooksack River than in the other two sites, having the highest nutrient concentrations, greatest discharge, and largest values for Q/w. Relative nutrient attenuation at the North Fork Skokomish River and Thornton Creek are comparable throughout the year despite being different types of streams (high gradient headwater stream compared with low gradient urban stream, respectively). The monthly mean nutrient concentrations from North Fork Skokomish River were much lower than from Thornton Creek (table 8); however, monthly mean discharge at Thornton Creek was much lower than at North Fork Skokomish River (table 7). The similarity in relative attenuation at these two sites shows how the physical (hydrologic) and biological conditions in streams and rivers are important, and the interaction of these factors can improve attenuation potential even in highly developed watersheds.

Future Work

There are several ways forward for improving understanding of nutrient attenuation in Puget Sound stream and rivers. Firstly, a simple procedure was developed to score the relative nutrient attenuation potential of a given stream or river reach for NO_3^- and PO_4^{-3} based on four factors (v_f, channel slope, channel sinuosity, and specific discharge [Q/w]). The scoring procedure showed that for two different attenuation models, higher scores represent higher relative attenuation values. Therefore, compilation of more detailed spatial data sets for these factors will allow for a better broad-scale assessment of attenuation potential in the Puget Sound Basin. Much of this data likely are already being collected as part of other programs by Washington State Department of Ecology and other local and federal agencies, but these data need to be assembled. Particularly, channel slope, sinuosity, and Q/w, or a subset of these data are likely collected during stream habitat assessments and estimates of channel slope and sinuosity can be determined from high-resolution GIS coverages. Seasonal changes in these factors also could be collected and summarized and used to further understand and assess temporal changes in attenuation.

One other parameter that could help managers understand nutrient attenuation dynamics in the region is the amount of fine benthic organic matter contained within streams and rivers. It has been shown that the amount of fine benthic organic matter is important for NO_3^- removal (Mulholland and others, 2000; Meyer and others, 2005; Inwood and others, 2007; Harvey and others, 2013); however, detailed spatial data on this parameter (percent fines) for Puget Sound streams and rivers are not currently available. For this reason, this parameter was not included in the scoring procedure even though is it an important factor. If this data were spatially available, scoring for orthophosphorus attenuation may be improved because this parameter indicates the extent of sediment deposition, and thus deposition of sediment bound P. These data, if collected during instream habitat assessments, could help explain controls on instream nutrient attenuation in the future.

A large data gap exists for Puget Sound Basin streams and rivers. Information on v_f is not currently available for Puget Sound. Estimates of v_f based on a literature-based correlation between nutrient concentration and v_f from other studies were used for this report; however, the procedure used to estimate nutrient concentration was based on modeled data and involved a complicated procedure to calculate reach-scale concentrations. A first step in developing a Puget Sound Basin dataset for v_f would involve compiling mean annual and (or) monthly nutrient concentrations for monitoring locations to calculate v_f using equations 8 and 9. Other efforts to develop a v_f dataset for Puget Sound include conducting field experiments to measure v_f directly across a range of nutrient concentrations and stream types in order to derive a specific correlation between concentration and v_f for the Puget Sound Basin. These data are best collected using stream nutrient injections, but this approach is better for use on small to medium sized reaches and can be labor-intensive. For large main stem reaches, if groundwater and hyporheic exchange loads are assumed negligible, calculating upstream-downstream loads also can be used to estimate v_f.

No studies could be found that directly measured nutrient attenuation in streams and rivers in the Puget Sound Basin. Existing ambient monitoring networks do not typically include multiple stations along a river reach; however, longitudinal sampling of rivers and streams can provide detailed information on spatial and temporal attenuation. Additionally, longitudinal data may be collected under other programs (for example, during total maximum daily load assessments) and could be compiled in order to begin estimating attenuation at specific locations around Puget Sound. Beyond conducting detailed field assessments of instream attenuation, the PNW SPARROW model (Wise and Johnson, 2013) could be recalibrated for the Puget Sound Basin and used to determine nutrient attenuation dynamics in the region. In addition to redefining the PNW SPARROW model for the Puget Sound Basin, the expression used to describe instream attenuation can be updated to use the v_f model as was done in Aguilera and others (2013). The current PNW SPARROW model uses an expression to describe instream loss based on physical factors only and as shown in this report, attenuation is related to physical and biological metrics.

Emerging techniques using continuous NO_3^- monitors may allow for detailed analysis of longitudinal estimates of attenuation or time-series estimates of attenuation at a fixed groundwater-streamgage station. For example, Hensley and others (2014) used continuous NO_3^- monitors that recorded data as they were dragged behind a small watercraft to produce detailed longitudinal profiles for NO_3^-. From these data, sections of the NO_3^- curve were used to calculate attenuation metrics (S_w, U, and v_f) in great spatial detail. Continuous NO_3^- monitors also can be coupled with a new generation of USGS groundwater-streamgages to infer real-time NO_3^- attenuation at fixed stations. This technique combines the instrumentation of existing streamgages with real-time groundwater-level data from an adjacent shallow well to provide detailed information on surface water-groundwater interactions (Eddy-Miller and others, 2012). The data on groundwater-surface water interactions can be coupled with the continuous NO_3^- data (collected in a stream) and continuous or discrete NO_3^- data collected from the shallow well to estimate nutrient transport though the stream bottom (Barlow and Coupe, 2012).

Overall, through compiling new and old data in Puget Sound Basin streams and rivers for various channel characteristics (Q/w, slope, sinuosity, substrate size) and other chemical and biological factors (nutrient concentrations and v_f data) and integrating the collection of new data with an appreciation for the types of data needed to characterize attenuation, a better understanding of the spatial and temporal dynamics of instream nutrient attenuation can be achieved. Little is known about the incremental effect that the physical, chemical, and biological factors presented in this report have on overall nutrient attenuation, and future research is needed to understand the synergistic effects of how multiple attenuation factors influence stream and river nutrient removal.

Management of Instream Nutrient Attenuation

Based on the information presented in this report, several actions are available for managing instream nutrient loads and attenuation of nutrient loads in stream and rivers of the Puget Sound Basin. Firstly, it is important to preserve areas that currently indicate high potential for instream nutrient attenuation, namely the small streams in headwaters of river basins. Secondly, work to improve reaches that are currently degraded and show a low potential for nutrient attenuation. These areas include smaller reaches in developed areas and larger main stem reaches. It is not feasible to make large main stem reaches into small headwater reaches, but restoration techniques can be used to try and mimic small stream features in larger reaches.

Generally, preservation and improvement of instream nutrient attenuation should focus on two factors: (1) increasing the travel time through a reach and contact time of water sediment (reactive) surfaces and (2) lowering nutrient concentrations (and loads) to avoid saturation of instream attenuation and increase attenuation efficiency. Increasing the travel time and contact time with sediment surface in a reach can be achieved by maintaining or reestablishing channel-flood plain connectivity and instream channel complexity features that promote groundwater-surface water exchange (woody debris dams, riffle-pool morphology, side channels, and meandering reaches). Maintaining and restoring channel-flood plain connectivity will help disperse floods, increasing overall travel times through the reach and reducing the proportion of flow that remains in the main channel, thus lowering Q/w. In turn, restoration of flood plain connectivity can be achieved by increasing levy setbacks or removing channelization features entirely. Restoration projects that include addition of woody debris, constructed step-pools maintaining heterogeneous substrate, aquatic planting, and creating sinuosity in previously channelized reaches will increase the potential for hyporheic exchange and the contact time for biological uptake of nutrients and lead to greater nutrient uptake (Shibata and others 2004; Kasahara and Hill 2006a, 2006b).

To address the second factor, management of point and non-point sources of nutrients and maintaining or re-establishing riparian vegetation along reaches will work to lower nutrient concentrations and instream loads. Intact riparian zones have been shown to greatly reduce nutrient delivery to streams and rivers through uptake in soils and vegetation from overland and groundwater flowpaths to the stream reach (Naiman and Décamps, 1997). River-flood plain connectivity also has the benefit of increasing potential for nutrient attenuation when floodwater is dispersed to areas that are optimal for the removal of nutrients thus reducing nutrient burden for downstream reaches (Sheibley and others, 2006).

Land use affects both the hydrology and biology of streams and rivers through modification of flow regimes from urbanization and increased nutrient delivery from urban and agricultural practices. Although urban and suburban development is inevitable, effects from altered flow regimes (higher leak flows, flashier hydrographs) and higher nutrient loads may be mitigated by preserving riparian zones along streams and the implementation of low impact development practices.

Key management goals that can be used to maintain and improve instream attenuation include:

1. Preserve areas where relative nutrient attenuation is high, namely small headwater reaches by maintaining important channel features (woody debris, instream vegetation, step-pool morphology, channel sinuosity) and preserve intact riparian zones.

2. Restore channels where relative attenuation is low, large main stem reaches, and smaller reaches with high nutrient inputs.

Specific actions that can be utilized to reach these goals include:

1. Restore instream habitat through addition of woody debris, heterogeneous substrate (gravel and cobbles), and constructed step-pools.
2. Restore and maintain channel-flood plain connectivity through increased levy setback or levy removal.
3. Restore and maintain healthy riparian zones along streams.
4. Reduce point and non-point source nutrient loads.
5. Where development cannot be avoided, mitigate potential degradation of channels by retaining intact riparian zones and through implementation of low impact development.

As these management actions are undertaken, an appreciation of the data needs and modeling tools presented can be used to quantify the nutrient attenuation success of these efforts. Many of these management approaches are already being undertaken during projects aimed to restore quality salmon habitat throughout the Puget Sound Basin. Therefore, there is a dual benefit to these projects that also may lead to enhanced potential for N and P attenuation.

Summary and Conclusions

A review of the scientific literature identified several physical, chemical, and biological factors that influence nutrient retention in streams and rivers. A number of models were identified that can be used to estimate reach scale attenuation. Attenuation is dependent on both physical and biological factors, and the importance of these factors changes with location in the watershed. For example, attenuation is generally high in headwater reaches compared to main stem reaches likely because of a combination of long travel times and low nutrient concentrations, and more biological activity. In contrast, main stem reaches tend to have lower attenuation potential because they have shorter travel times because these channels are larger, water moves faster than in headwater reaches, and therefore have less contact with bed sediments. Relative to headwater reaches, main stem reaches generally tend to have higher nutrient concentrations resulting from adjacent developed land uses and more human activities. Main stem reaches also have been shown to have low biological diversity due to other water-quality issues, and biomonitoring of these lowland reaches shows that many of them are degraded. Attenuation of orthophosphate (PO_4^{-3}) is much lower than nitrate (NO_3^-) across Puget Sound Basin streams and rivers. Because relative attenuation in the larger main stem reaches is lower does not mean attenuation is not important, rather these larger reaches have a lower potential for nutrient removal than smaller headwater reaches.

Two models of relative NO_3^- attenuation based on physical factors (RivR-N) and physical and biological factors (v_f model) were compared to identify where these factors are more important in the Puget Sound Basin. In headwater reaches, the v_f model simulates much higher relative attenuation indicating that biological factors in these small channels may be more important. In contrast, in lower gradient, large channels of the main stem reaches, the RivR-N model simulates greater relative attenuation indicating that hydrologic (physical) factors may play a bigger role on instream attenuation of NO_3^-. Comparing these two models is not without its limitations because the assumptions and error associated with each model are likely different. However, this comparison does begin to indicate where physical and biological factors may be important and warrants further investigation.

Some limitations of the two models (RivR-N and v_f model) used for the analysis in this report that should be mentioned. Both the attenuation models were applied to a refined stream and river network that did not include the smallest watersheds (greater than 50 square kilometers); therefore, the results do not include the smaller headwater reaches. This approach was used because there was accurate and detailed data on channel characteristics (widths, slopes, and sinuosity) for the refined river network; therefore, attenuation could be described accurately on a spatial basis at this resolution. For the RivR-N model, the expression used to estimate instream attenuation was based on field studies outside the Puget Sound region and does not include attention data from the scientific literature from the last decade. Despite this, the model did use the best available information at that time and data used to derive equation 4 included a wide variety of stream and river types. Our application of the RivR-N model relied on estimates of reach width based on correlation between mean annual discharge and width (equation 5) at locations of USGS streamgages. Because streamgages are in areas with stable, straight channels the attenuation is expected to be underestimated at unconfined reaches using this approach. For the v_f model, the addition of a biological factor (v_f) results in additional limitations and assumptions in our approach. Modeled concentration data were relied on for total nitrogen (TN) and total phosphorus (TP), correlations between total and dissolved nitrogen and phosphorus, and expressions for v_f in terms of concentration (equation 7 and 8) in order to parameterize this model. Although the approach used for estimating reach-scale v_f is acceptable and uses the available data, the v_f model would be more accurate with real Puget Sound Basin data for v_f. The v_f model assumes that dispersion and transient storage in the reach is negligible. This may be true in some reaches, but these assumptions would not be universally met throughout the Puget Sound Basin.

The v_f model for PO_4^{-3} attenuation may underestimate total attenuation because it is based mainly on the biological uptake of phosphorus and does not describe loss by sedimentation well. This may be one reason that the scoring procedure did not work as well for PO_4^{-3} attenuation. Despite the limitation of these models, for the initial analysis of attenuation in the Puget Sound Basin where the focus was on general patterns, these two models described similar patterns of instream nutrient attenuation that are consistent with current understanding of the factors that influence attenuation in streams and rivers.

Four primary factors were identified that accurately describe the patterns observed from the RivR-N and v_f models; the uptake velocity (v_f), channel sinuosity, channel slope, and specific discharge (Q/w). These factors incorporate the strength of biological activity of a reach, the duration of time that water spends in a reach, and the amount of contact between surface water and the streambed. Criteria derived in this report allows for a scoring of a stream or river reach in order to identify the degree of potential attenuation. However, without detailed reach-specific data, and the Puget Sound Basin as a whole, this approach should be considered a preliminary, first level analysis. A first step for future investigations is to compile detailed datasets for these four factors and improve the scoring procedure presented in this report.

Temporal variability of relative attenuation was examined in detail at three case study locations by varying flow and nutrient concentrations. Results indicate that relative attenuation may reach a minimum when NO_3^- concentrations approach about 3 milligrams nitrogen per liter (mg N/L) during low-flow conditions and at about 1 mg N/L during high-flow conditions. For PO_4^{-3}, relative attenuation is minimized at much lower concentrations (about 0.1 milligram phosphorus per liter [mg P/L]) for low- and high-flow conditions. These results indicate the need to keep instream concentrations low in order to avoid reaching minimum attenuation, which will lead to increased transport of nutrients to downstream reaches. This can be done through the reduction of both point and nonpoint sources of nutrients to stream and rivers, particularly in larger main stem reaches where relative attenuation is less. Preliminary seasonal analysis at three case study sites showed that attenuation is highest in summer months and lowest in winter and spring. Summer months typically correspond to lower flows and higher biological activity. Variability in physical factors (discharge and channel width) resulted in greater monthly variability in attenuation than did changes in monthly nutrient concentrations. The analysis of seasonal relative attenuation also showed that the headwater reaches of the North Fork Skokomish River and highly urban Thornton Creek have comparable levels of relative attention throughout the year. The similarity in relative attenuation at these two sites shows how both the physical (hydrologic) and biological conditions in streams and rivers are important and the interaction of these factors can improve attenuation potential even in highly developed watersheds.

References Cited

Aguilera, R., Marcé, R., and Sabater, S., 2013, Modeling nutrient retention at the watershed scale—Does small stream research apply to the whole river network?: Journal of Geophysical Research—Biogeosciences, v. 118, p. 728–740.

Ahearn, D.S., Sheibley, R.W. , Dahlgren, R.A., and Keller, K.E., 2004, Temporal dynamics of stream water chemistry in the last free-flowing river draining the western Sierra Nevada, California: Journal of Hydrology, v. 295, p. 47–63.

Alexander, R.B., Bohlke, J.K. Boyer, E.W., David, M.B., Harvey, J.W., Mulholland, P.J., and others, 2009, Dynamic modeling of nitrogen losses in river networks unravels the coupled effects of hydrological and biogeochemical processes: Biogeochemistry, v. 93, p. 91–116, doi:10.1007/s10533-008-9274-8.

Alexander, R.B., Boyer, E.W., Smith, R.A., Schwarz, G.E., and Moore R.B., 2007, The role of headwater streams in downstream water quality: Journal of the American Water Resources Association, v. 43, p. 41–59.

Alexander, R.B., Smith, R.A., and Schwarz, G.E., 2000, Effect of stream channel size on the delivery of nitrogen to the Gulf of Mexico: Nature, v. 403 p. 758–761.

Allan, D.J., and Castillo, M.M., 2007, Stream ecology—Structure and function of running waters (2d ed.): Springer, 436 p.

Arheimer, B., and Brandt, M., 1998, Modelling nitrogen transport and retention in the catchments of southern Sweden: Ambio, v. 27, p. 471–480.

Baker, D.W., Bledsoe, B.P., and Price, J.M., 2012, Stream nitrate uptake and transient storage over a gradient of geomorphic complexity, north-central Colorado, USA: Hydrological Processes, v. 26, p. 3,241–3,252, doi:10.1002/hyp.8385.

Barlow, J.R.B., and Coupe, R.H., 2012, Groundwater and surface-water exchange and resulting nitrate dynamics in the Bogue Phalia Basin in northwestern Mississippi: Journal of Environmental Quality, v. 41, p. 155–169.

Bechtold, J.S., Edwards, R.T., and Naiman, R.J., 2003, Biotic versus hydrologic control over seasonal nitrate leaching in a floodplain forest: Biogeochemistry, v. 63, p. 53–71.

Beechie, T.J., Liermann, M., Pollock, M.M., Baker S., and Davies J., 2006, Channel pattern and river-floodplain dynamics in forested mountain river systems: Geomorphology, v. 78, p. 124–141.

Behrendt, H., and Opitz, D., 1999, Retention of nutrients in river systems—Dependence on specific runoff and hydraulic load: Hydrobiologia, v. 410, p. 111–122.

Bellmore, J.R., and Baxter, C.V., 2013, Effects of geomorphic process domains on river ecosystems—A comparison of floodplain and confined valley segments: River Research and Applications, v. 30, no. 5, p. 617-630, doi:10.1002/rra.2672.

Bencala, K.E., 1983, Simulation of solute transport in a mountain pool-and-riffle stream with a kinetic mass transfer model for sorption: Water Resources Research, v. 19, p. 732–738.

Bencala, K.E., and Walters, R.A., 1983, Simulation of solute transport in a mountain pool-and-riffle stream—A transient storage model: Water Resources Research, v. 19, p. 718–724.

Bernot, M.J., and Dodds, W.K., 2005, Nitrogen retention, removal, and saturation in lotic ecosystems: Ecosystems, v. 8, p. 442–453.

Bernot, M.J., Tank, J.L., Royer, T.V., and David, M.B., 2006, Nutrient uptake in streams draining agricultural catchments of the Midwestern United States: Freshwater Biology, v. 51, p. 499–509.

Black, R.W., Munn, M.D., and Plotnikoff, R.W., 2004, Using macroinvertebrates to identify biota-land cover optima at multiple scales in the Pacific Northwest, USA: Journal of the North American Benthological Society, v. 23, p. 340–362.

Black, R.W., and Silkey M., 1998, Water-quality assessment of the Puget Sound Basin, Washington—Summary of stream biological data through 1995: U.S. Geological Survey Water-Resources Investigations Report 97-4164, 78 p.

Boano, F., Camporeale, C., Revelli, R., and Ridolfi, L., 2006, Sinuosity-driven hyporheic exchange in meandering rivers: Geophysical Research Letters, v. 33, L18406, doi:10.1029/2006GL027630.

Boano, F., Harvey, J.W., Marion, A., Packman, A.I., Revelli, R., Ridolfi, L., and others, 2014, Hyporheic flow and transport processes—Mechanisms, models, and biogeochemical implications: Reviews of Geophysics, v. 52, no. 4, p. 603–679, doi:10.1002/2012rg000417.

Booth, D.B., Hartley, D., and Jackson R., 2002, Forest cover, impervious-surface area, and the mitigation of stormwater impacts: Journal of the American Water Resources Association, v. 38, p. 835–845.

Brett, M.T., Arhonditsis, G.B., Mueller, S.E., Hartley, D.M., Frodge, J.D., and Funke, D.E., 2005a, Non-point-source impacts on stream nutrient concentrations along a forest to urban gradient: Environmental Management, v. 35, p. 330–342.

Brett, M.T., Mueller, S.E., and Arhonditsis, G.B., 2005b, A daily time series analysis of stream water phosphorus concentrations along an urban to forest gradient: Environmental Management, v. 35, p. 56–71.

Briggs, M.A., Gooseff, M.N., Arp, C.D., and Baker, M.A., 2009, A method for estimating surface transient storage parameters for streams with concurrent hyporheic storage: Water Resources Research, v. 45, W00D27, doi:10.1029/2008wr006959.

Collins, B.D., and Montgomery, D.R., 2002, Forest development, wood jams and restoration of floodplain rivers in the Puget Lowland: Restoration Ecology, v. 10, p. 237–247.

Collins, B.D., Montgomery, D.R., Fetherston, K.L., and Abbe, T.B., 2012, The floodplain large-wood cycle hypothesis—A mechanism for the physical and biotic structuring of temperate forested alluvial valleys in the North Pacific coastal ecoregion: Geomorphology, v. 139–140, p. 460–470.

Collins, B.D., Montgomery, D.R., and Haas, A.D., 2002, Historical changes in the distribution and functions of large wood in Puget Lowland rivers: Canadian Journal of Fisheries and Aquatic Sciences, v. 59, p. 66–76.

Conn, K.E., and Black, R.W., 2014, Data compilation for assessing sediment and toxic chemical loads from the Green River to the lower Duwamish Waterway, Washington: U.S. Geological Survey Data Series 880, 46 p., http://dx.doi.org/10.3133/ds880.

Covino, T., McGlynn, B., and Baker, M., 2010, Separating physical and biological nutrient retention and quantifying uptake kinetics from ambient to saturation in successive mountain stream reaches: Journal of Geophysical Research, v. 115, G04010, doi:10.1029/2009JG001263.

Darracq, A., and Destouni, G., 2005, In-stream nitrogen attenuation—Model-aggregation effects and implications for coastal nitrogen impacts: Environmental Science and Technology, v. 39, p. 3716–3722.

Davidsson, T.E., and Ståhl, M., 2000, The influence of organic carbon on nitrogen transformations in five wetland soils; Soil Science Society of America Journal, v. 64, p. 1129–1136.

Davis, J.C., and Minshall, G.W., 1999, Nitrogen and phosphorus uptake in two Idaho (USA) headwater wilderness streams: Oecologia, v. 119, p. 247–255.

Dodds, W.K., Beaulieu, J.J., Eichmiller, J.J., Fischer, J.R., Franssen, N.R., Gudder, D.A., and others, 2008, Nitrogen cycling and metabolism in the thalweg of a prairie river: Journal of Geophysical Research, v. 113, G04029, doi:doi:10.1029/2008JG000696.

Dodds, W.K., Lopez, A.J., Bowden, W.B., Gregory, S., Grimm, N.B., Hamilton, S.K., and others, 2002, N uptake as a function of concentration in streams: Journal of the North American Benthological Society, v. 21, p. 206–220.

Donner, S.D., Kucharik, C.J., and Oppenheimer, M., 2004, The influence of climate on in-stream removal of nitrogen: Geophysical Research Letters, v. 31, L20509, p. 20501–20505.

Doyle, M.W., 2005, Incorporating hydrologic variability into nutrient spiraling: Journal of Geophysical Research, v. 110, no. G1, doi:10.1029/2005jg000015.

Dubrovsky, N.M., Burow, K.R., Clark, G.M., Gronberg, J.M., Hamilton P.A., Hitt, K.J., Mueller, D.K., Munn, M.D., Nolan, B.T., Puckett, L.J., Rupert, M.G., Short, T.M., Spahr, N.E., Sprague, L.A., and Wilber, W.G., 2010, The quality of our Nation's waters—Nutrients in the Nation's streams and groundwater, 1992–2004: U.S. Geological Survey Circular 1350, 174 p.

Duff, J.H., Hendricks, S.P., Jackman, A.P., and Triska, F.J., 2002, The effect of *Elodea canadensis* beds on pore water chemistry, microbial respiration, and nutrient retention in the Shingobee River, Minnesota, North America: Verhandlungen Internationalen Vereiningen Limnologie, v. 28, p. 214–222.

Duff, J.H., Tesoriero, A.J., Richardson, W.B., Strauss, E.A., and Munn, M.D., 2008, Whole-stream response to nitrate loading in three streams draining agricultural landscapes: Journal of Environmental Quality, v. 37, p. 1133–1144.

Duff, J. H., and Triska, F.J., 1990, Denitrification in sediments from the hyporheic zone adjacent to a small forested stream: Canadian Journal of Fisheries and Aquatic Sciences, v. 47, p. 1140–1147.

Duff, J.H., and Triska, F.J., 2000, Nitrogen biogeochemistry and surface-subsurface exchange in streams *in* Jones, J.B., and Mulholland, P.J., eds., Streams and ground waters: San Diego, Academic Press, p. 197–220.

Earl, S.R., Valett, H.M., and Webster, J.R., 2007, Nitrogen spiraling in streams—Comparisons between stable isotope tracer and nutrient addition experiments: Limnology and Oceanography, v. 52, p. 1718–1723.

Ebbert, J.C., Embrey, S.S., Black, R.W., Tesoriero, A.J., and Haggland, A.L., 2000, Water quality in the Puget Sound Basin, Washington and British Columbia, 1996–98: U.S. Geological Survey Circular 1216, 31 p.

Eddy-Miller, C.A., Constantz, J., and Wheeler, J.D., 2012, Demonstrating usefulness of real-time monitoring at streambank wells coupled with active streamgages—Pilot studies in Wyoming, Montana, and Mississippi: U.S. Geological Survey Fact Sheet 2012-3054, 6 p.

Elliott, A.H., and Brooks, N.H., 1997, Transfer of nonsorbing solutes to a streambed with bed forms—Laboratory experiments: Water Resources Research, v. 33, no. 1, p. 137–151, doi:10.1029/96WR02783.

Embrey, S.S., and Frans, L.M., 2003, Surface-water of the Skokomish, Nooksack, and Green-Duwamish Rivers and Thornton Creek, Puget Sound Basin, Washington, 1995–98: U.S. Geological Survey Water-Resources Investigations Report 02–4190, 192 p.

Embrey, S.S., and Inkpen, E.L., 1998, Water-quality assessment of the Puget Sound Basin, Washington, nutrient transport in rivers, 1980–93: U.S. Geological Survey Water-Resources Investigations Report 97–4270, 30 p.

Ensign, S.H., and Doyle, M.W., 2005, In-channel transient storage and associated nutrient retention—Evidence from experimental manipulations: Limnology and Oceanography, v. 50, p. 1740–1751.

Ensign, S.H., and Doyle, M.W., 2006, Nutrient spiraling in streams and river networks: Journal of Geophysical Research—Biogeosciences, v. 111, G04009.

Fellows, C.S., Valett, H.M., Dahm, C.N., Mulholland, P.J., and Thomas, S.A., 2006, Coupling nutrient uptake and energy flow in headwater streams: Ecosystems, v. 9, p. 788–804.

Fry, J., Xian, G., Jin, S., Dewitz, J., Homer, C., Yang, L., Barnes, C., Herold, N., and Wickham, J., 2011, Completion of the 2006 National Land Cover Database for the conterminous United States: Photogrammetric Engineering and Remote Sensing, v. 77, no. 9, p. 858–864.

Garcia-Ruiz, R., Pattinson, S.H., and Whitton, B.A., 1998, Kinetic parameters of denitrification in a river continuum: Applied and Environmental Microbiology, v. 64, p. 2533–2538.

Gooseff, M.N., Anderson, J.K., Wondzell, S.M., LaNier, J., and Haggerty, R., 2006: A modelling study of hyporheic exchange pattern and the sequence, size, and spacing of stream bedforms in mountain stream networks, Oregon, USA: Hydrological Processes, v. 20, p. 2443–2457.

Gooseff, M.N., Hall, R.O., Jr., and Tank, J.L., 2007, Relating transient storage to channel complexity in streams of varying land use in Jackson Hole, Wyoming: Water Resources Research, v. 43, W01417, doi:01410.01029/02005WR004626.

Grimm, N.B., and Petrone, K.C., 1997, Nitrogen fixation in a desert stream ecosystem: Biogeochemistry, v. 37, p. 33–61.

Hall, R.O., Jr., Bernhardt, E.S., and Likens, G.E., 2002, Relating nutrient uptake with transient storage in forested mountain streams: Limnology and Oceanography, v. 47, p. 255–265.

Hall, R.O., Jr., and Tank J.L., 2003, Ecosystem metabolism controls nitrogen uptake in streams in Grand Teton National Park, Wyoming: Limnology and Oceanography, v. 48, p. 1120–1128.

Hall, R.O., Jr., Tank, J.L., Sobota, D.J., Mulholland, P.J., O’Brien, J.M., Dodds, W.K., and others, 2009, Nitrate removal in stream ecosystems measured by 15N addition experiments—Total uptake: Limnology and Oceanography, v. 54, p. 653–665.

Hanafi, S., Grace, M.R., and Hart, B.T., 2006, Can nutrient spiralling be used to detect seasonal nutrient uptake in a forested stream?: Water, Air, and Soil Pollution—Focus, v. 6, p. 403–411.

Harvey, J.W., Böhlke, J.K., Voytek, M.A., Scott D., and Tobias, C.R., 2013, Hyporheic zone denitrification—Controls on effective reaction depth and contribution to whole-stream mass balance: Water Resources Research, v. 49, p. 6298–6316, doi:10.1002/wrcr.20492.

Harvey, J.W., Conklin, M.H., and Koelsch, R.S., 2003, Predicting changes in hydrologic retention in an evolving semi-arid alluvial stream: Advances in Water Resources, v. 26, p. 939–950.

Harvey, J.W., Wagner, B.J., and Bencala, K.E., 1996, Evaluating the reliability of the stream tracer approach to characterize stream-subsurface water exchange: Water Resources Research, v. 32, p. 2441–2451.

Helton, A.M., Poole, G.C., Meyer, J.L., Wollheim, W.M., Peterson, B.J., Mulholland, P.J., and others, 2011, Thinking outside the channel—Modeling nitrogen cycling in networked river ecosystems: Frontiers in Ecology and the Environment, v. 9, p. 229–238, doi:10.1890/080211.

Hendricks, S.P. and White, D.S., 1995, Seasonal biogeochemical patterns in surface water, subsurface hyporheic, and riparian ground water in a temperate stream ecosystem: Archiv fur Hydrobiologie, v. 134, p. 459–490.

Hendricks, S.P. and White D.S., 2000, Stream and groundwater influences on phosphorus biogeochemistry, *in* Jones, J.B., and Mulholland, P.J., eds., Streams and ground waters: San Diego, Academic Press, p. 222–235.

Hensley, R.T., Cohen, M.J., and Korhnak, L.V., 2014, Inferring nitrogen removal in large rivers from high-resolution longitudinal profiling: Limnology and Oceanography, v. 59, p. 1152–1170.

Herrera Environmental Consultants, Inc., 2007, Water quality statistical and pollutant loading analysis—Green-Duwamish water quality assessment: Herrera Environmental Consultants, prepared for King County Department of Natural Resources and Parks, Seattle, Washington.

Herrera Environmental Consultants, Inc., 2011, Toxics in surface runoff to Puget Sound—Phase 3 Data and Load estimates: Prepared for Washington State Department of Ecology, Olympia, Publication No. 11-03-010, 257 p.

Hester, E.T., Young, K.I., and Widdowson, M.A., 2013, Mixing of surface and groundwater induced by riverbed dunes—Implications for hyporheic zone definitions and pollutant reactions: Water Resources Research, v. 49, p. 5221–5237, doi:10.1002/wrcr.20399.

Holloway, J.M., Dahlgren, R.A., Hansen, B., and Casey, W.H., 1998, Contribution of bedrock nitrogen to high nitrate concentrations in stream water: Nature, v. 395, p. 785–788.

Holmes, R.M., Fisher, S.G., and Grimm, N.B., 1994, Parafluvial nitrogen dynamics in a desert stream ecosystem: Journal of the North American Benthological Society, v. 13, p. 468–478.

Horizon Systems Corporation, 2012, National Hydrography Dataset Plus version 2: Horizon Systems Corporation dataset, accessed April, 25, 2014 at http://www.horizon-systems.com/NHDPlus/NHDPlusV2_data.php.

Howarth, R.W., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., and others, 1996, Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean—Natural and human influences: Biogeochemistry, v. 35, p. 75–139.

Inkpen, E.L., and Embrey, S.S., 1998, Nutrient transport in the major rivers and streams of the Puget Sound Basin, Washington: U.S. Geological Survey Fact Sheet 009–98, 4 p.

Inwood, S.E., Tank, J.L., and Bernot, M.J., 2005, Patterns of denitrification associated with land use in 9 midwestern headwater streams; Journal of the North American Benthological Society, v. 24, p. 227–245.

Inwood, S.E., Tank, J.L., and Bernot, M.J., 2007, Factors controlling sediment denitrification in midwestern streams of varying land use: Microbial Ecology, v. 53, p. 247–258.

Jansson, M., Andersson, R., Berggren, H., and Leonardson, L., 1994, Wetlands and lakes as nitrogen traps: Ambio, v. 23, p. 320–325.

Jones, J.B. and Mulholland, P.J., eds., 2000, Streams and ground waters: San Diego, Academic Press, 425 p.

Jung, H.S., and Deng, Z.Q., 2011, Modeling of nitrogen retention in Amite River: Water, Air, and Soil Pollution, v. 215, p. 411–425.

Kasahara, T., and Hill., A.R., 2006a, Effects of riffle-step restoration on hyporheic zone chemistry in N-rich lowland streams: Canadian Journal of Fisheries and Aquatic Sciences, v. 63, p. 120–133.

Kasahara, T., and Hill, A.R., 2006b, Hyporheic exchange flows induced by constructed riffles and steps in lowland streams in southern Ontario, Canada: Hydrological Processes, v. 20, p. 4287–4305.

Kelly, C.A., Rudd, J.W.M., Hesslein, R.H., Schindler, D.W., Dillon, P.J., Driscoll, C.T., Gherini, S.A., and Hecky, R.E., 1987, Prediction of biological acid neutralization in acid-sensitive lakes: Biogeochemistry, v. 3, p. 129–141.

Kemp, M.J., and Dodds, W.K., 2002, The influence of ammonium, nitrate, and dissolved oxygen concentrations on uptake, nitrification, and denitrification rates associated with prairie stream substrata: Limnology and Oceanography, v. 47, p. 1380–1393.

Konrad, C.P., Booth, D.B., and Burges, S.J., 2005, Effects of urban development in the Puget Lowland, Washington, on interannual streamflow patterns—Consequences for channel form and streambed disturbance: Water Resources Research, v. 41, p. 1–15.

Konrad, C.P., 2015, Geospatial assessment of ecological functions and flood-related risks on floodplains along major rivers in the Puget Sound Basin, Washington: U.S. Geological Survey Scientific Investigations Report 2015–5033, 28 p., http://dx.doi.org/10.3133/sir20155033.

Knighton, D., 1998, Fluvial forms and processes: London, Arnold Publishers, 383 p.

Kunza, L.A., and Hall, R.O., Jr., 2014. Nitrogen fixation can exceed inorganic nitrogen uptake fluxes in oligotrophic streams: Biogeochemistry, v. 121, p. 537–549.

Lautz, L.K., and Siegel, D.I., 2007, The effect of transient storage on nitrate uptake lengths in streams—An inter-site comparison: Hydrological Processes, v. 21, p. 3533–3548.

Macrae, M.L., English, M.C., Schiff, S.L., and Stone, M., 2007, Capturing temporal variability for estimates of annual hydrochemical export from a first-order agricultural catchment in southern Ontario, Canada: Hydrological Processes, v. 21, p. 1651–1663.

Magirl, C.S., and Olsen, T.D., 2009, Navigability potential of Washington rivers and streams determined with hydraulic geometry and a geographic information system: U.S. Geological Survey Scientific Investigations Report 2009-5122, 22 p.

Martin, L.A., Mulholland, P.J., Webster, J.R., and Valett, H.M., 2001, Denitrification potential in sediments of headwater streams in the southern Appalachian Mountains, USA: Journal of the North American Benthological Society, v. 20, p. 505–519.

May, C.W., Horner, R.R., Karr, J.R., Mar, B.W., and Welch, E.B., 1997, Effects of urbanization on small streams in the Puget Sound Ecoregion: Watershed Protection Techniques, v. 2, no. 4, p. 483–494.

McBride, M., and Booth, D.B., 2005, Urban impacts on physical stream condition—Effects of spatial scale, connectivity, and longitudinal trends: Journal of the American Water Resources Association, v. 41, p. 565–580.

Merritt, G., and Hartman, C., 2009, Status of Puget Sound tributaries 2009: Olympia, Washington State Department of Ecology, Biology, Chemistry, and Physical Habitat. Environmental Assessment Program, Publication No. 12-03-029.

Meyer, J.L., Paul, M.J., and Taulbee, W.K., 2005, Stream ecosystem function in urbanizing landscapes. Journal of the North American Benthological Society, v. 24, p. 602–612.

Mitsch, W.J., Horne, A.J., and Nairn, R.W., 2000, Nitrogen and phosphorus retention in floodplains—Ecological approaches to solving excess nutrient problems: Ecological Engineering, v. 14, p. 1–7.

Mohamedali, T., Roberts, M., Sackmann, B., and Kolosseus, A., 2011, Puget Sound dissolved oxygen model nutrient load summary for 1999–2008: Olympia, Washington State Department of Ecology Publication No. 11-03-001.

Morley, S.A., and Karr, J.R., 2002, Assessing and restoring the health of urban streams in the Puget Sound Basin: Conservation Biology, v. 16, p. 1498–1509.

Mulholland, P.J., Best, G.R., Coutant, C.C., Hornberger, G.M., Meyer, J.L., Robinson, P.J., and others, 1997, Effects of climate change on freshwater ecosystems of the south-eastern United States and the Gulf Coast of Mexico: Hydrological Processes, v. 11, p. 949–970.

Mulholland, P.J., Fellows, C.S., Tank, J.L., Grimm, N.B., Webster, J.R., Hamilton, S.K., and others, 2001, Inter-biome comparison of factors controlling stream metabolism: Freshwater Biology, v. 46, p. 1503–1517.

Mulholland, P.J., Hall, R.O., Jr., Sobota, D.J., Dodds, W.K., Findlay, S.E.G., Grimm, N.B., and others, 2009, Nitrate removal in stream ecosystems measured by 15N addition experiments—Denitrification: Limnology and Oceanography, v. 54, p. 666–680.

Mulholland, P.J., Helton, A.M., Poole, G.C., Hall, R.O., Jr., Hamilton, S.K., Peterson, B.J., and others, 2008, Stream denitrification across biomes and its response to anthropogenic nitrate loading: Nature, v. 452, p. 202–205.

Mulholland, P.J., Steinman, A.D., and Elwood, J.W., 1990, Measurement of phosphorus uptake length in streams—Comparison of radiotracer and stable PO4 releases: Canadian Journal of Fisheries and Aquatic Sciences: v. 47, p. 2351–2357.

Mulholland, P.J., Steinman, A.D., Marzolf, E.R., Hart, D.R., and DeAngelis, D.L., 1994, Effect of periphyton biomass on hydraulic characteristics and nutrient cycling in streams: Oecologia, v. 98, p. 40–47.

Mulholland , P.J., Tank, J.L., Sanzone, D.M., Webster, J.R., Wollheim, W., Peterson, B.J., and others, 2000, Ammonium and nitrate uptake lengths in a small forested stream determined by 15N tracer and short-term nutrient enrichment experiments: Verhandlungen Internationalen Vereiningen Limnologie, v. 27, p. 1320–1325.

Mulholland, P.J., Thomas, S.A., Valett, H.M., Webster, J.R., and Beaulieu, J., 2006, Effects of light on NO_3^- uptake in small forested streams—Diurnal and day-to-day variations: Journal of the North American Benthological Society, v. 25, p. 583–595.

Naiman, R.J., and Décamps, H., 1997, The ecology of interfaces—Riparian zones: Annual Review of Ecology and Systematics, v. 28, p. 621–658.

Natho, S. and Venohr, M., 2014, Active versus potential floodplains-the effect of small flood events on nutrient retention along the river Elbe corridor (Germany): Aquatic Sciences, v. 76, no. 4, p. 633–642.

Newbold, J.D., Elwood, J.W., O'Neill, J.W., and Van Winkle, W., 1981, Measuring nutrient spiralling in streams: Canadian Journal of Fisheries and Aquatic Sciences, v. 38, p. 860–863.

Olde Venterink, H., Hummelink, E., and Van Den Hoorn, M.W., 2003, Denitrification potential of a river floodplain during flooding with nitrate-rich water—Grasslands versus reedbeds: Biogeochemistry, v. 65, p. 233–244.

Payn, R.A., Webster, J.R., Mulholland, P.J., Maurice Valett, H., and Dodds, W.K., 2005, Estimation of stream nutrient uptake from nutrient addition experiments: Limnology and Oceanography—Methods 3, p. 174–182.

Pinay, G., Gumiero, B., Tabacchi, E., Gimenez, O., Tabacchi-Planty, A.M., Hefting, M.M., and others, 2007, Patterns of denitrification rates in European alluvial soils under various hydrological regimes: Freshwater Biology, v. 52, p. 252–266.

Rabalais, N.N., Turner, R.E., Justic, D., Dortch, Q., Wiseman, W.J., Jr., and Sen Gupta, B.K., 1996, Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf: Estuaries, v. 19, p. 386–407.

Ranalli, A.J., and Macalady, D.L., 2010, The importance of the riparian zone and in-stream processes in nitrate attenuation in undisturbed and agricultural watersheds—A review of the scientific literature; Journal of Hydrology, v. 389, p. 406–415, doi:10.1016/j.jhydrol.2010.05.045.

Reddy, K.R., Kadlec, R.H., Flaig, E., and Gale. P.M., 1999, Phosphorus retention in streams and wetlands—A Review: Critical Reviews in Environmental Science and Technology, v. 29, no. 1, p. 83–146.

Roberts, B.J., and Mulholland, P.J., 2007, In-stream biotic control on nutrient biogeochemistry in a forested headwater stream, West Fork of Walker Branch: Journal of Geophysical Research—Biogeosciences, v. 112, G04002, doi:doi:10.1029/2007JG000422.

Roberts, M., Mohamedali, T., and Sackmann, B., 2014, Puget Sound and the straits dissolved oxygen assessment—Impacts of current and future nitrogen sources and climate change through 2070: Olympia, Washington State Department of Ecology Publication No. 14-03-007.

Roberts, M. L., and Bilby, R.E., 2009, Urbanization alters litterfall rates and nutrient inputs to small Puget Lowland streams: Journal of the North American Benthological Society, v. 28, p. 941–954.

Roberts, M.L., Bilby, R.E., and Booth, D.B., 2008, Hydraulic dispersion and reach-averaged velocity as indicators of enhanced organic matter transport in small Puget Lowland streams across an urban gradient: Fundamental and Applied Limnology, v. 171, p. 145–159.

Sand-Jensen, K., and Mebus, J.R., 1996, Fine-scale patterns of water velocity within macrophyte patches in Danish streams; Oikos, v. 76, p. 169–180.

Saunders, D.L., and Kalff, J., 2001, Nitrogen retention in wetlands, lakes and rivers; Hydrobiologia, v. 443, p. 205–212.

Seitzinger, S.P., Styles, R.V., Boyer, E.W., Alexander, R.B., Billen, G., Howarth, R.W., and others, 2002, Nitrogen retention in rivers—Model development and application to watersheds in the northeastern USA: Biogeochemistry, v. 57–58, p. 199–237.

Sheibley, R.W., Ahearn, D.S., and Dahlgren, R.A., 2006, Nitrate loss from a restored floodplain in the Lower Cosumnes River, California: Hydrobiologia, v. 571, p. 261–272.

Sheibley, R.W., Duff, J.H., Jackman, A.P., and Triska, F.J., 2003a, Inorganic nitrogen transformations in the bed of the Shingobee River, Minnesota—Integrating hydrologic and biological processes using sediment perfusion cores: Limnology And Oceanography, v. 48, p. 1129.

Sheibley, R.W., Duff, J.H., and Tesoriero, A.J., 2014, Low transient storage and uptake efficiencies in seven agricultural streams—Implications for nutrient demand: Journal of Environmental Quality, v. 43, p. 1980–1990, doi:10.2134/jeq2014.01.0034.

Sheibley, R.W., Jackman, A.P., Duff, J.H., and Triska, F.J., 2003b, Numerical modeling of coupled nitrification-denitrification in sediment perfusion cores from the hyporheic zone of the Shingobee River, MN: Advances in Water Resources, v. 26, no. 9, p. 977–987.

Shibata, H., Kuraji, K., Toda, H., and Sasa, K., 2001, Regional comparison of nitrogen export to Japanese forest streams; The Scientific World, v. 1, p. 572–580.

Shibata, H., Sugawara, O., Toyoshima, H., Wondzell, S.M., Nakamura, F. Kasahara, T., and others, 2004, Nitrogen dynamics in the hyporheic zone of a forested stream during a small storm, Hokkaido, Japan: Biogeochemistry, v. 69, p. 83–103.

Simon, K.S., Townsend, C.R., Biggs, B.J.F., and Bowden, W.B., 2005, Temporal variation of N and P uptake in 2 New Zealand streams: Journal of the North American Benthological Society, v. 24, p. 1–18.

Smith, R.A., Schwarz, G.E., and Alexander, R.B., 1997, Regional interpretation of water-quality monitoring data: Water Resources Research, v. 33, p. 2781–2798.

Staubitz, W.W., Bortleson, G.C., Semans, S.D., Tesoriero, A.J., and Black, R.W., 1997, Water-quality assessment of the Puget Sound Basin, Washington—Environmental setting and its implications for water quality and aquatic biota: U.S. Geological Survey Water-Resources Investigations Report 97–4013, 76 p.

Steinberg, P.D., Brett, M.T., Bechtold, J.S., Richey, J.E., Porensky, L.M., and Smith, S.N., 2011, The influence of watershed characteristics on nitrogen export to and marine fate in Hood Canal, Washington, USA: Biogeochemistry, v. 106, p. 415–433.

Stonedahl, S.H., Harvey, J.W. and Packman, A.I., 2013, Interactions between hyporheic flow produced by stream meanders, bars, and dunes: Water Resources Research, v. 49, p. 5450–5461, doi:10.1002/wrcr.20400.

Strauss, E. A., Richardson, W.B., Bartsch, L.A., Cavanaugh, J.C., Bruesewitz, D.A., Imker, H., Heinz, J.A., and Soballe, D.M., 2004, Nitrification in the Upper Mississippi River—Patterns, controls, and contribution to the NO_3^- budget: Journal of the North American Benthological Society, v. 23, p. 1–14.

Stream Solute Workshop, 1990, Concepts and methods for assessing solute dynamics in stream ecosystems: Journal of the North American Benthological Society, v. 9, p. 95–112.

Tank, J.L., Rosi-Marshall, E.J., Baker, M.A., and Hall, R.O., Jr., 2008, Are rivers just big streams?—A pulse method to quantify nitrogen demand in a large river: Ecology, v. 89, p. 2935–2945.

Thibodeaux, L.J., and Boyle, J.D., 1987, Bedform-generated convective transport in bottom sediment: Nature, v. 325, p. 341–343.

Tockner, K., Pennetzdorfer, D., Reiner, N., Schiemer F., and Ward, J.V., 1999, Hydrological connectivity, and the exchange of organic matter and nutrients in a dynamic river-floodplain system (Danube, Austria): Freshwater Biology, v. 41, p. 521–535.

Tonina, D., and Buffington, J.M., 2007, Hyporheic exchange in gravel bed rivers with pool-riffle morphology—Laboratory experiments and three-dimensional modeling: Water Resources Research, v. 43, p. 1–16.

Tonina, D., and Buffington, J.M., 2011, Effects of stream discharge, alluvial depth and bar amplitude on hyporheic flow in pool-riffle channels: Water Resources Research, v. 47, W08508, doi:10.1029/2010wr009140.

Triska, F.J., Duff, J.H., Sheibley, R.W., Jackman, A.P., and Avanzino, R.J., 2007, DIN retention-transport through four hydrologically connected zones in a headwater catchment of the Upper Mississippi River: Journal of the American Water Resources Association, v. 43, p. 60–71.

Triska, F.J., Kennedy, V.C., Avanzino, R.J., Zellweger, G.W., and Bencala, K.E., 1989, Retention and transport of nutrients in a third-order stream in northwestern California—Hyporheic Processes: Ecology, v. 70, p. 1893–1905.

Triska, F., Pringle, C.M., Duff, J.H., Avanzino, R.J., and Zellweger, G., 2006, Soluble reactive phosphorus (SRP) transport and retention in tropical, rain forest streams draining a volcanic landscape in Costa Rica—In situ SRP amendment to streams and laboratory studies: Biogeochemistry, v. 81, p. 145–157.

U.S. Environmental Protection Agency, 1986, Quality criteria for water—1986: Environmental Protection Agency Report, EPA 440/5-86-002 (variously paged).

U.S. Geological Survey, 2012, National Elevation Dataset, 10 m: U.S. Geological Survey dataset, accessed March 11, 2014, at http://ned.usgs.gov/.

U.S. Geological Survey, 2013, National Hydrography Dataset—high resolution hydrography: U.S. Geological Survey dataset, accessed April 25, 2014, at ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution/NHDH_WA_931v210.zip.

Valett, H.M., Crenshaw, C.L., and Wagner, P.F., 2002, Stream nutrient uptake, forest succession, and biogeochemical theory: Ecology, v. 83, p. 2888–2901.

Vaux, W.G., 1968, Intragravel flow and interchange of water in a streambed: Fishery Bulletin, v. 66, p 479–489.

Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., and others, 1997, Human alteration of the global nitrogen cycle—Sources and consequences: Ecological Applications, v. 7, p. 737–750.

Washington Department of Ecology, 2013a, National Estuary Program toxics and nutrients—Federal fiscal year 2013 work plan and multi-year implementation strategy: Washington Department of Ecology, accessed December 20, 2014, at http://www.ecy.wa.gov/puget_sound/docs/grants/2012fedgrants_NEP_multiyearstrategy.pdf.

Washington Department of Ecology, 2013b, Puget Sound watershed characterization: Washington Department of Ecology, accessed May 17, 2014, at http://www.ecy.wa.gov/services/gis/data/pugetsound/characterization.htm.

Webster, J.R., Mulholland, P.J., Tank, J.L., Valett, H.M., Dodds, W.K., Peterson, B.J., and others, 2003, Factors affecting ammonium uptake in streams—An inter-biome perspective: Freshwater Biology, v. 48, p. 1329–1352.

Webster, J.R., and Patten, B.C., 1979, Effects of watershed perturbation on stream potassium and calcium dynamics: Ecological Monographs, v. 49, p. 51–72.

White D.S., and Hendricks, S.P., 2000, Lotic macrophytes and surface-subsurface exchange processes, *in* Jones, J.B., and Mulholland, P.J., eds., Streams and ground waters: San Diego, Academic Press, p. 363–379.

Wise, D.R., and Johnson, H.M., 2011, Surface-water nutrient conditions and sources in the United States Pacific Northwest: Journal of the American Water Resources Association, v. 47, no. 5, p. 1–26, doi:10.1111/j.1752-1688.2011.00580.x.

Wise, D.R., and Johnson, H.M., 2013, Application of the SPARROW model to assess surface-water nutrient conditions and sources in the United States Pacific Northwest: U.S. Geological Survey Scientific Investigations Report 2013–5103, 32 p.

Withers, P.J.A., and Jarvie, H.P., 2008, Delivery and cycling of phosphorus in rivers—A review: Science of the Total Environment, v. 400, p. 379–395.

Wollheim, W.M., Vörösmarty, C.J., Peterson, B.J., Seitzinger, S.P., and Hopkinson, C.S., 2006, Relationship between river size and nutrient removal: Geophysical Research Letters, v. 33, L06410, doi:10.1029/2006GL025845.

Appendix A. Relationship Between Mean Annual Discharge and Drainage Area for Puget Sound Rivers and Streams

Table A1. Summary of flow data used in figure 6 for calculating mean annual discharge at the 535 rivers segments of the hydrologic network in this report.

[**Abbreviations:** km^2, square kilometer; m^3/s, cubic meter per second]

Site No.	Site name	Drainage area (km^2)	Years of record	Water year		Mean annual discharge (m^3/s)
				Start	End	
12200500	Skagit River near Mount Vernon	7,918	30	1981	2010	462.3
12194000	Skagit River near Concrete	7,010	30	1981	2010	418.8
12035002	Chehalis River near Satsop	4,508	6	2003	2008	175.5
12150800	Snohomish River near Monroe	3,935	30	1981	2010	263.8
12181000	Skagit River at Marblemount	3,535	30	1981	2010	171.4
12031000	Chehalis River at Porter	3,313	30	1981	2010	112.6
12179000	Skagit River above Alma Creek near Marblemount	3,261	15	1981	1995	142.0
12178000	Skagit River at Newhalem	3,008	30	1981	2010	123.1
12101500	Puyallup River at Puyallup	2,427	30	1981	2010	90.79
12027500	Chehalis River near Grand Mound	2,291	30	1981	2010	78.70
12213100	Nooksack River at Ferndale	2,012	29	1981	2010	108.8
12189500	Sauk River near Sauk	1,828	30	1981	2010	121.4
12149000	Snoqualmie River near Carnation	1,544	30	1981	2010	100.2
12210700	Nooksack River at North Cedarville	1,508	5	2006	2010	108.3
12210500	Nooksack River at Deming	1,495	25	1981	2005	94.38
12134500	Skykomish River near Gold Bar	1,370	30	1981	2010	111.6
12089500	Nisqually River at Mckenna	1,324	30	1981	2010	34.50
12100496	White River near Auburn	1,216	18	1988	2009	25.87
12040500	Queets River near Clearwater	1,139	30	1981	2010	128.5
12100000	White River at Buckley	1,093	22	1981	2003	16.48
12099200	White River above Boise Creek at Buckley	1,052	7	2004	2010	35.44
12098500	White River near Buckley	1,027	23	1981	2003	39.75
12113000	Green River near Auburn	1,021	30	1981	2010	36.75
12097850	White River below Clearwater River near Buckley	960	15	1983	2010	36.44
12144500	Snoqualmie River near Snoqualmie	960	30	1981	2010	73.26
12035000	Satsop River near Satsop	765.0	30	1981	2010	59.03
12193500	Baker River at Concrete	760.0	28	1981	2008	74.24
12086500	Nisqually River at La Grande	747.5	30	1981	2010	40.71
12045500	Elwha River at McDonald Bridge near Port Angeles	688.6	30	1981	2010	42.76
12039500	Quinault River at Quinault Lake	675.8	30	1981	2010	83.06
12167000	North Fork Stillaguamish River near Arlington	670.7	30	1981	2010	54.08
12041200	Hoh River at U.S. Highway 101 near Forks	647.7	30	1981	2010	72.49
12106700	Green River at Purification Plant near Palmer	591.4	30	1981	2010	25.46
12061500	Skokomish River near Potlatch	581.1	29	1981	2009	35.28
12105900	Green River below Howard A. Hanson Dam	565.8	30	1981	2010	26.89
12017000	North River near Raymond	560.6	5	1996	2000	32.48
12044900	Elwha River above Lake Mills near Port Angeles	506.9	9	1995	2010	37.08
12119000	Cedar River at Renton	471.0	30	1981	2010	17.78
12093500	Puyallup River near Orting	440.3	30	1981	2010	20.60
12080010	Deschutes River at E St. Bridge at Tumwater	414.7	20	1991	2010	11.10
12125200	Sammamish River near Woodinville	407.0	25	1981	2005	8.327
12048000	Dungeness River near Sequim	399.4	30	1981	2010	11.00
12025000	Newaukum River near Chehalis	396.8	29	1981	2010	14.33
12037400	Wynoochee River above Black Creek near Montesano	396.8	29	1981	2010	35.72
12141300	Middle Fork Snoqualmie River near Tanner	394.2	29	1981	2010	34.04

Table A1. Summary of flow data used in figure 6 for calculating mean annual discharge at the 535 rivers segments of the hydrologic network in this report.—Continued

[**Abbreviations:** km^2, square kilometer; m^3/s, cubic meter per second]

Site No.	Site name	Drainage area (km^2)	Years of record	Water year Start	Water year End	Mean annual discharge (m^3/s)
12186000	Sauk River above Whitechuck River near Darrington	389.1	30	1981	2010	30.88
12082500	Nisqually River near National	340.5	30	1981	2010	21.59
12039005	Humptulips River below Highway 101 near Humptulips	337.9	8	2003	2010	39.48
12013500	Willapa River near Willapa	332.8	30	1981	2010	17.89
12043000	Calawah River near Forks	330.2	26	1985	2010	29.52
12155300	Pilchuck River near Snohomish	325.1	18	1993	2010	13.04
12117600	Cedar River below Diversion near Landsburg	317.4	18	1993	2010	14.63
12117500	Cedar River near Landsburg	309.8	30	1981	2010	18.52
12059500	North Fork Skokomish River near Potlatch	299.5	30	1981	2010	3.944
12020000	Chehalis River near Doty	289.3	30	1981	2010	16.30
12026400	Skookumchuck River near Bucoda	286.7	30	1981	2010	9.782
12175500	Thunder Creek near Newhalem	268.8	30	1981	2010	17.62
12205000	North Fork Nooksack River below Cascade Creek near Glacier	268.8	30	1981	2010	23.12
12091500	Chambers Creek below Leach Creek near Steilacoom	266.2	13	1998	2010	2.858
12209000	South Fork Nooksack River near Wickersham	263.7	13	1996	2008	19.27
12058800	North Fork Skokomish River below lower Cushman Dam near Potlatch	261.1	20	1989	2008	1.789
12104500	Green River near Lester	246.3	11	1981	1993	9.362
12138160	Sultan River below Powerplant near Sultan	241.2	27	1984	2010	20.93
12092000	Puyallup River near Electron	237.6	30	1981	2010	14.79
12079000	Deschutes River near Rainier	229.9	23	1981	2010	6.786
12201500	Samish River near Burlington	224.8	14	1997	2010	7.043
12116500	Cedar River at Cedar Falls	215.6	30	1981	2010	8.706
12116400	Cedar River at Powerplant at Cedar Falls	214.8	5	2006	2010	3.222
12144000	South Fork Snoqualmie River at North Bend	209.2	26	1985	2010	15.16
12148500	Tolt River near Carnation	208.4	30	1981	2010	15.12
12087000	Mashel River near La Grande	206.6	19	1992	2010	6.009
12095000	South Prairie Creek at South Prairie	203.5	23	1988	2010	6.244
12094000	Carbon River near Fairfax	202.0	19	1992	2010	12.07
12137800	Sultan River below Diversion Dam near Sultan	197.4	27	1984	2010	5.698
12060500	South Fork Skokomish River near Union	195.3	19	1981	2010	21.18
12083000	Mineral Creek near Mineral	192.5	21	1981	2006	10.30
12036000	Wynoochee River above Save Creek near Aberdeen	189.7	30	1981	2010	24.25
12039300	North Fork Quinault River near Amanda Park	189.7	6	1981	1986	25.38
12090500	Clover Creek near Tillicum	188.9	18	1991	2010	1.047
12097500	Greenwater River at Greenwater	188.2	17	1994	2010	5.146
12208000	Middle Fork Nooksack River near Deming	187.6	18	1993	2010	14.50
12112600	Big Soos Creek above Hatchery near Auburn	170.8	30	1981	2010	3.447
12054000	Duckabush River near Brinnon	170.2	30	1981	2010	11.86
12026150	Skookumchuck River below Bloody Run Creek near Centralia	168.7	29	1981	2010	7.456
12143600	South Fork Snoqualmie River at Edgewick	168.7	27	1984	2010	12.17
12142000	North Fork Snoqualmie River near Snoqualmie Falls	163.8	29	1981	2010	14.29
12056500	North Fork Skokomish River below Staircase Rapids near Hoodsport	146.4	30	1981	2010	14.91
12121600	Issaquah Creek near mouth near Issaquah	144.9	30	1981	2010	3.476
12076800	Goldsborough Creek above 7th Street at Shelton	140.5	6	2005	2010	4.314
12010000	Naselle River near Naselle	140.3	30	1981	2010	11.75
12043300	Hoko River near Sekiu	131.1	15	1996	2010	8.837

Table A1. Summary of flow data used in figure 6 for calculating mean annual discharge at the 535 rivers segments of the hydrologic network in this report—Continued

[**Abbreviations:** km², square kilometer; m³/s, cubic meter per second]

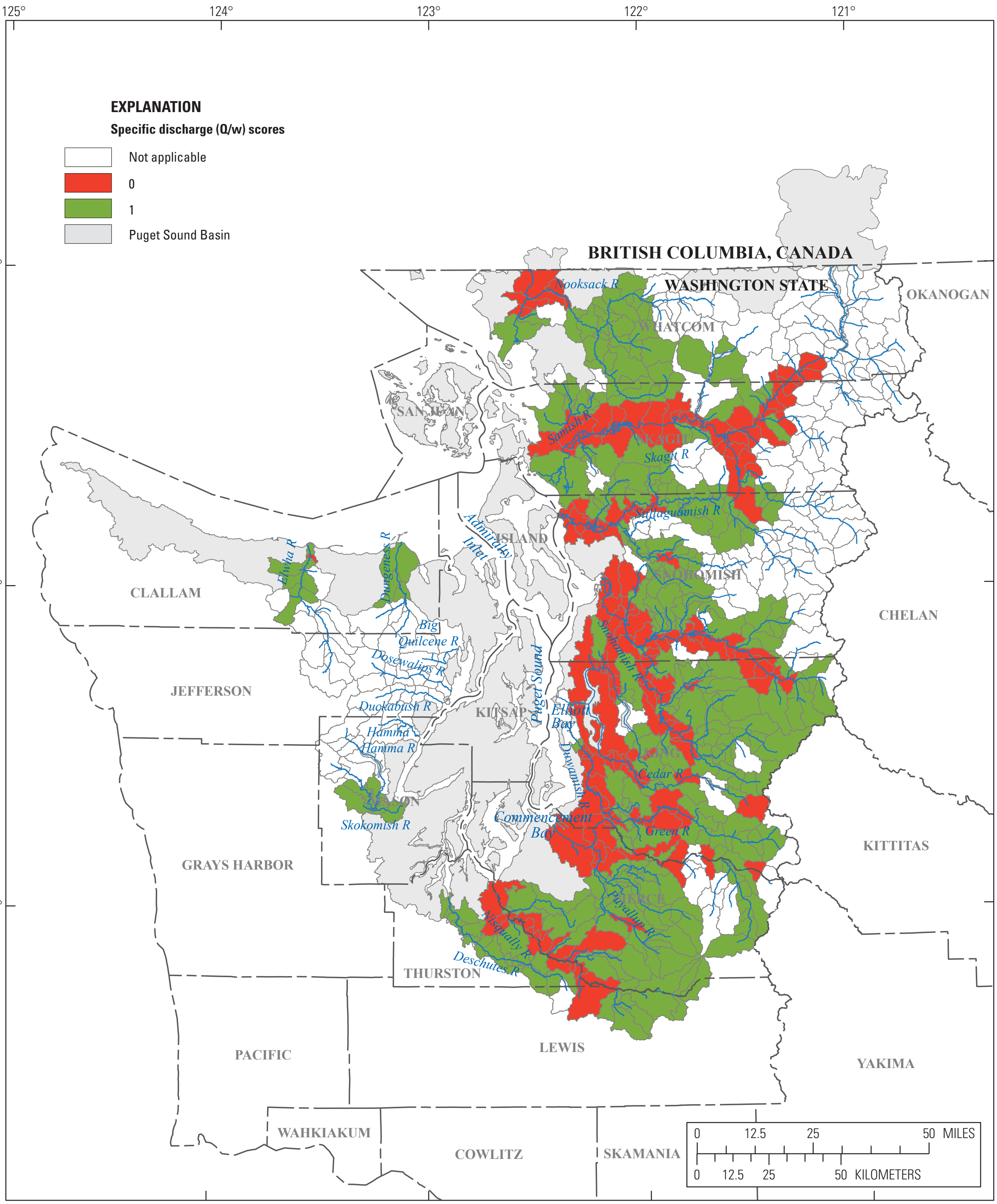
Site No.	Site name	Drainage area (km²)	Years of record	Water year		Mean annual discharge (m³/s)
				Start	End	
12179900	Bacon Creek below Oakes Creek near Marblemount	127.2	12	1999	2010	12.33
12052210	Big Quilcene River below Diversion near Quilcene	126.5	16	1995	2010	4.056
12143400	South Fork Snoqualmie River above Alice Creek near Garcia	106.5	29	1981	2010	8.234
12035400	Wynoochee River near Grisdale	105.7	30	1981	2010	14.57
12115000	Cedar River near Cedar Falls	104.2	30	1981	2010	6.824
12025700	Skookumchuck River near Vail	102.4	30	1981	2010	5.658
12147500	North Fork Tolt River near Carnation	102.1	30	1981	2010	9.750
12212050	Fishtrap Creek at Front Street at Lynden	96.77	12	1999	2010	1.994
12190710	Swift Creek near Concrete	93.06	8	1983	1990	11.76
12088000	Ohop Creek near Eatonville	88.32	17	1994	2010	1.785
12043163	Sooes River below Miller Creek near Ozette	81.92	6	1981	1986	6.271
12145500	Raging River near Fall City	78.34	30	1981	2010	3.510
12148300	South Fork Tolt River below regulating basin near Carnation	75.78	28	1983	2010	4.024
12108500	Newaukum Creek near Black Diamond	70.14	30	1981	2010	1.558
12178100	Newhalem Creek near Newhalem	68.86	30	1981	2010	4.989
12114500	Cedar River below Bear Creek near Cedar Falls	65.02	30	1981	2010	4.362
12127100	Swamp Creek at Kenmore	59.14	9	1981	1989	1.001
12209490	Skookum Creek above Diversion near Wickersham	58.88	12	1999	2010	3.994
12192700	Thunder Creek near Concrete	57.34	12	1983	1994	3.745
12148000	South Fork Tolt River near Carnation	50.43	30	1981	2010	2.722
12207850	Clearwater Creek near Welcome	47.36	10	1999	2010	3.328
12120600	Issaquah Creek near Hobart	45.06	23	1987	2010	1.326
12117000	Taylor Creek near Selleck	44.03	30	1981	2010	2.587
12105710	North Fork Green River near Lemolo	42.75	7	1981	1987	2.253
12103380	Green River above Twin Camp Creek near Lester	42.24	7	1993	1999	1.667
12099600	Boise Creek at Buckley	39.42	29	1981	2010	0.920
12158040	Tulalip Creek near Tulalip	39.42	9	2002	2010	0.349
12122500	Bear Creek near Redmond	35.58	8	1981	1996	0.615
12069550	Big Beef Creek near Seabeck	35.33	15	1981	2010	0.899
12115500	Rex River near Cedar Falls	34.30	30	1981	2010	2.698
12102075	Clarks Creek at Tacoma Road near Puyallup	33.28	13	1996	2008	1.626
12118500	Rock Creek near Maple Valley	32.26	8	2002	2010	0.381
12036400	Schafer Creek near Grisdale	30.98	10	1987	1996	2.143
12120000	Mercer Creek near Bellevue	30.72	30	1981	2010	0.640
12137290	South Fork Sultan River near Sultan	29.70	19	1992	2010	3.50
12118400	Rock Creek at Highway 516 near Ravensdale	28.67	9	2002	2010	0.313
12190718	Park Creek at Upper Bridge near Concrete	26.88	8	1983	1990	3.45
12206900	Racehorse Creek at North Fork Road near Kendall	26.88	12	1999	2010	1.66
12035450	Big Creek near Grisdale	24.50	16	1981	1996	3.15
12158010	Tulalip Creek above East Branch near Tulalip	24.17	9	2002	2010	0.199
12117820	Walsh Lake Ditch near Landsburg	24.12	5	1986	1990	0.320
12210900	Anderson Creek at Smith Road near Goshen	22.94	12	1999	2010	0.585
12113346	Springbrook Creek at Orillia	21.60	17	1994	2010	0.284
12157250	Mission Creek near Tulalip	20.28	9	2002	2010	0.151
12091100	Flett Creek at Tacoma	18.76	19	1981	2010	0.426
12120500	Juanita Creek near Kirkland	17.13	9	1981	1989	0.345
12091300	Leach Creek near Steilacoom	16.79	17	1981	2004	0.368
12073500	Huge Creek near Wauna	16.56	30	1981	2010	0.302

Table A1. Summary of flow data used in figure 6 for calculating mean annual discharge at the 535 rivers segments of the hydrologic network in this report.—Continued

[**Abbreviations:** km^2, square kilometer; m^3/s, cubic meter per second]

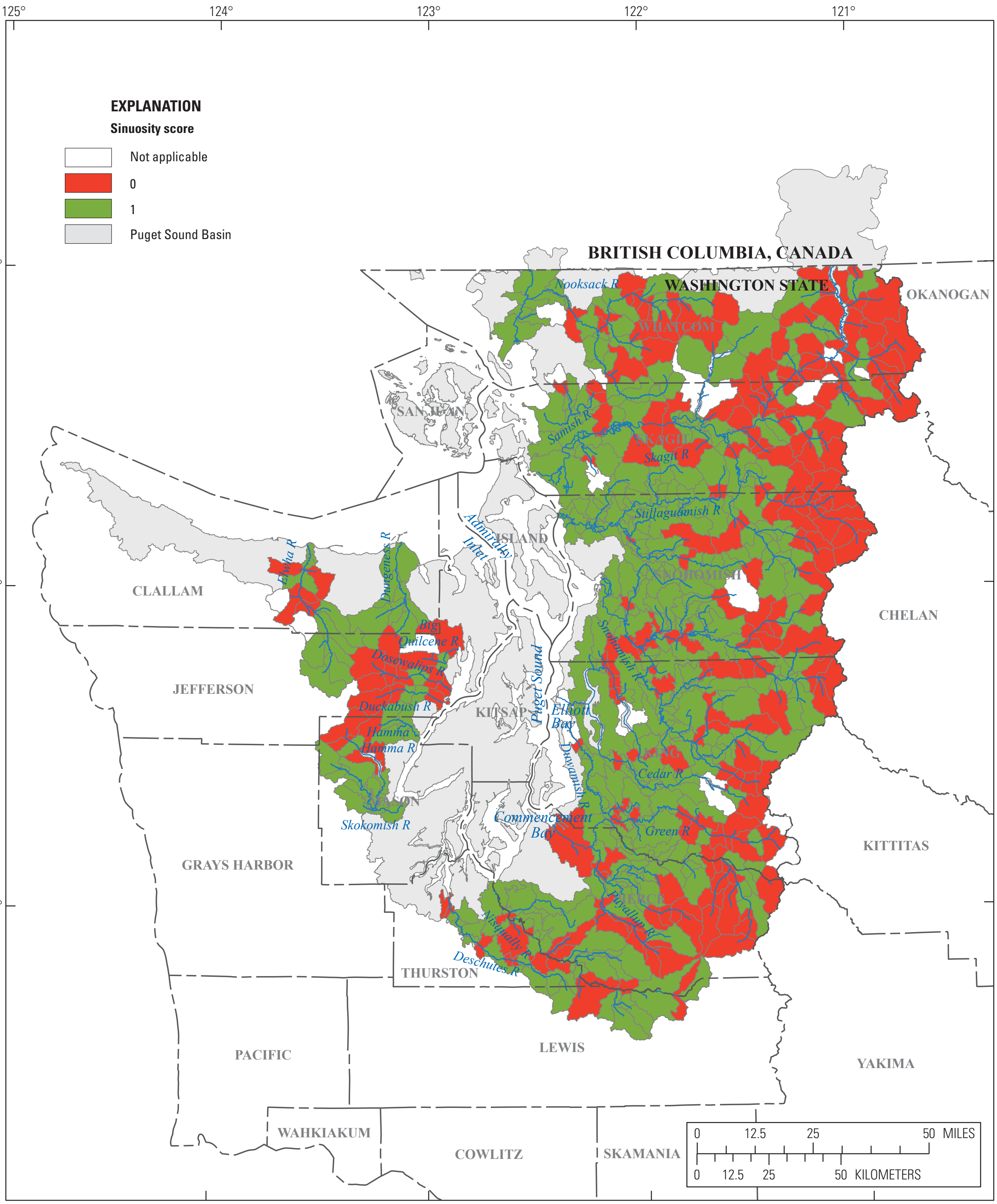
Site No.	Site name	Drainage area (km^2)	Years of record	Water year		Mean annual discharge (m^3/s)
				Start	End	
12090400	North Fork Clover Creek near Parkland	16.00	15	1992	2010	0.221
12113349	Mill Creek near mouth at Orillia	14.41	16	1995	2010	0.418
12147600	South Fork Tolt River near Index	13.67	30	1981	2010	1.547
12117700	Rock Creek above Walsh Lake Ditch near Landsburg	12.57	5	1986	1990	0.484
12091200	Leach Creek near Fircrest	12.11	27	1981	2010	0.164
12115700	Boulder Creek near Cedar Falls	11.88	26	1984	2010	0.687
12091180	Leach Creek at Holding Pond at Fircrest	11.75	5	1981	1985	0.141
12207750	Warm Creek near Welcome	10.57	11	1999	2009	0.777

Appendix B. Maps Showing Individual Scores for the Four Primary Factors Related to Nitrate and Orthophosphate Attenuation in Rivers and Streams in the Puget Sound Basin



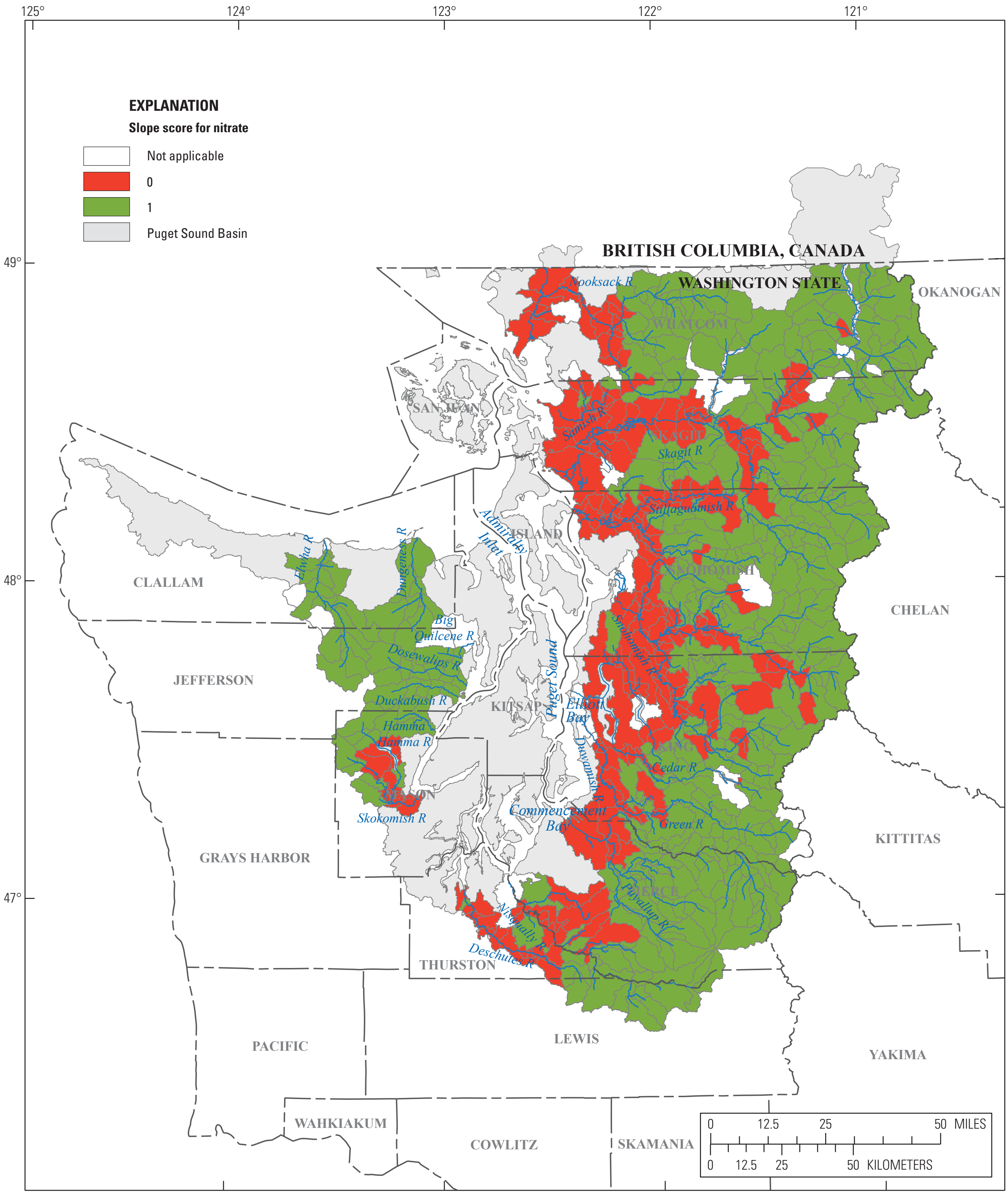
Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure B1. Specific discharge scores, Puget Sound Basin, Washington.



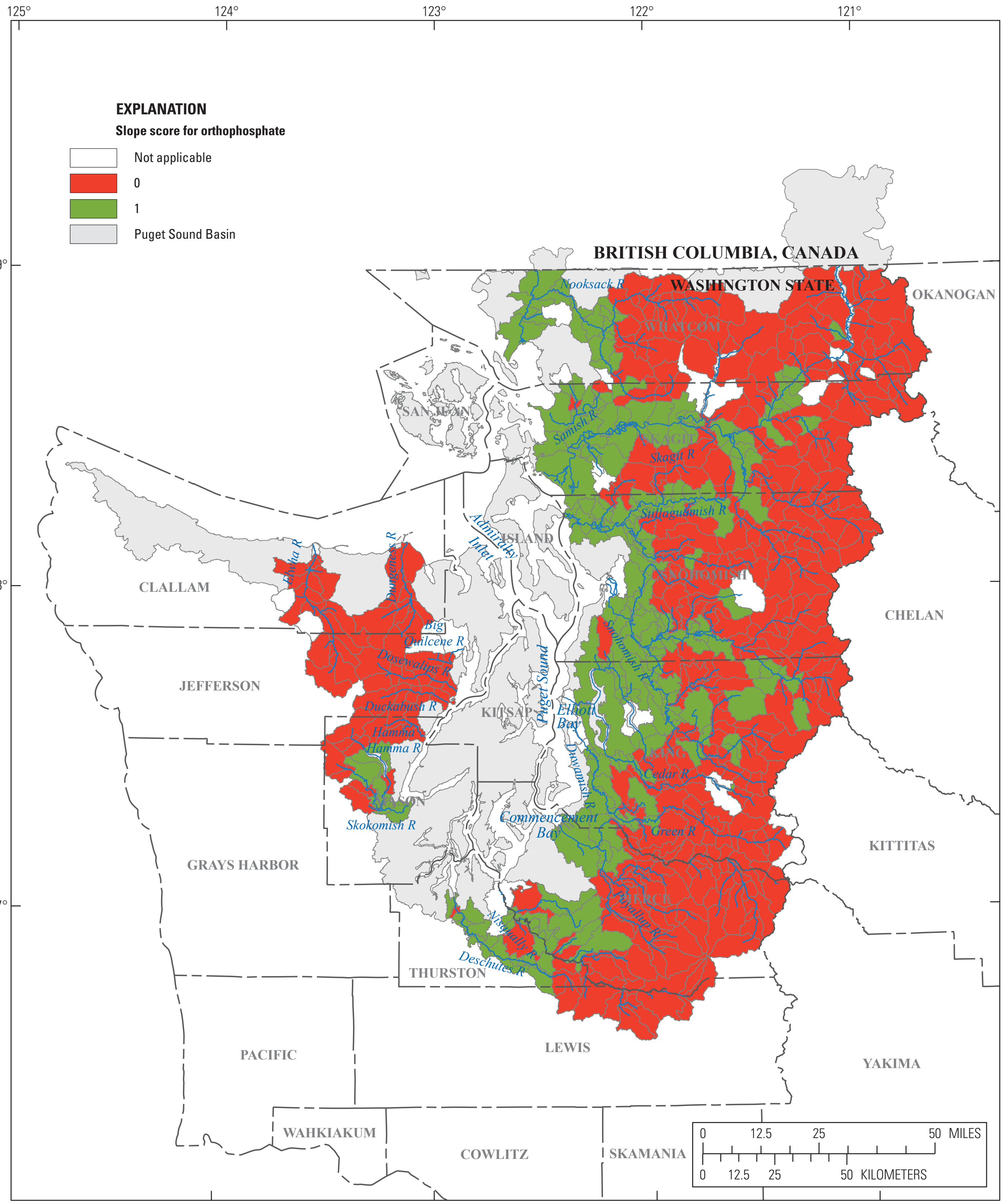
Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure B2. Sinuosity score, Puget Sound Basin, Washington.



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure B3. Slope score for nitrate, Puget Sound Basin, Washington.



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure B4. Slope score for orthophosphate, Puget Sound Basin, Washington.

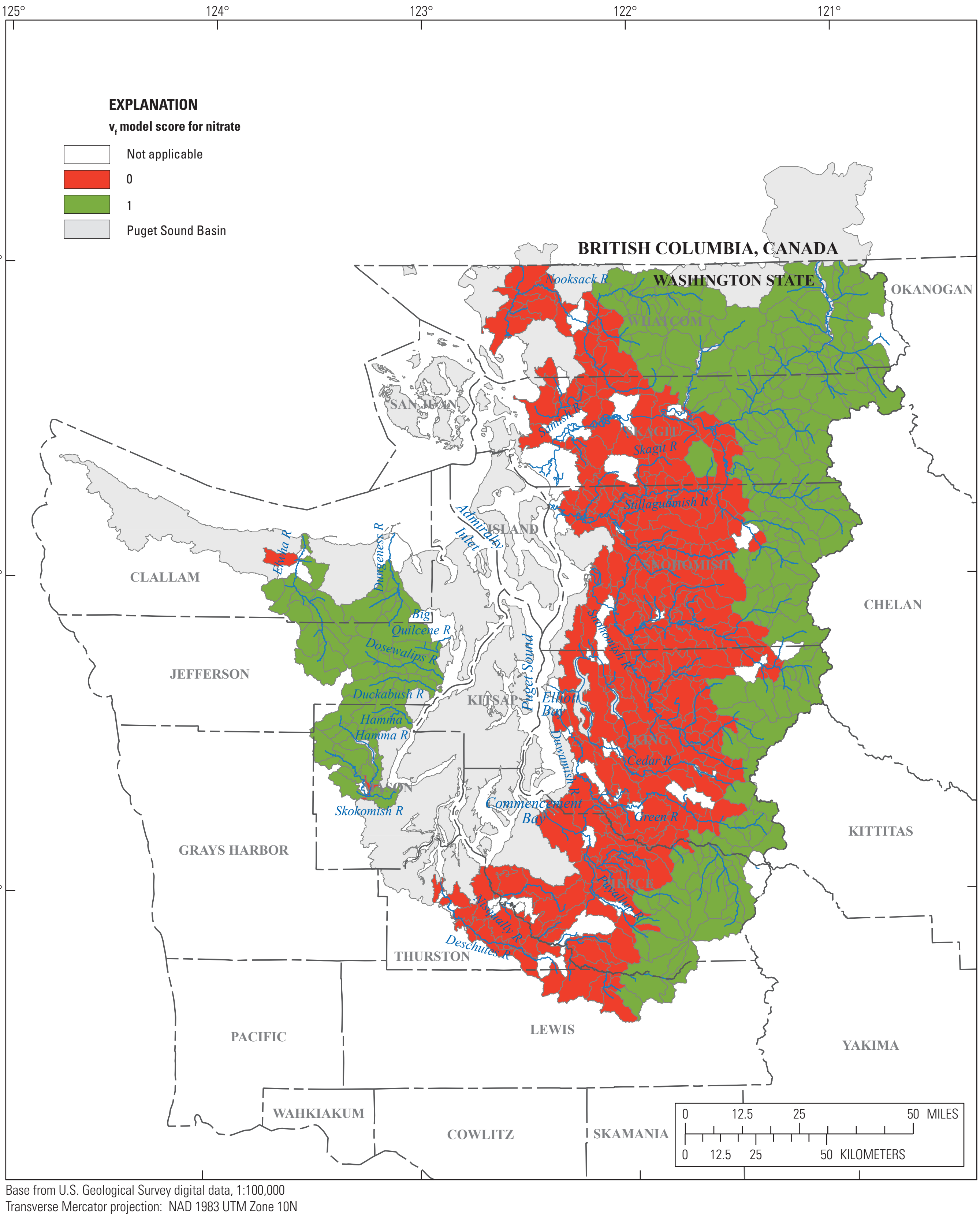
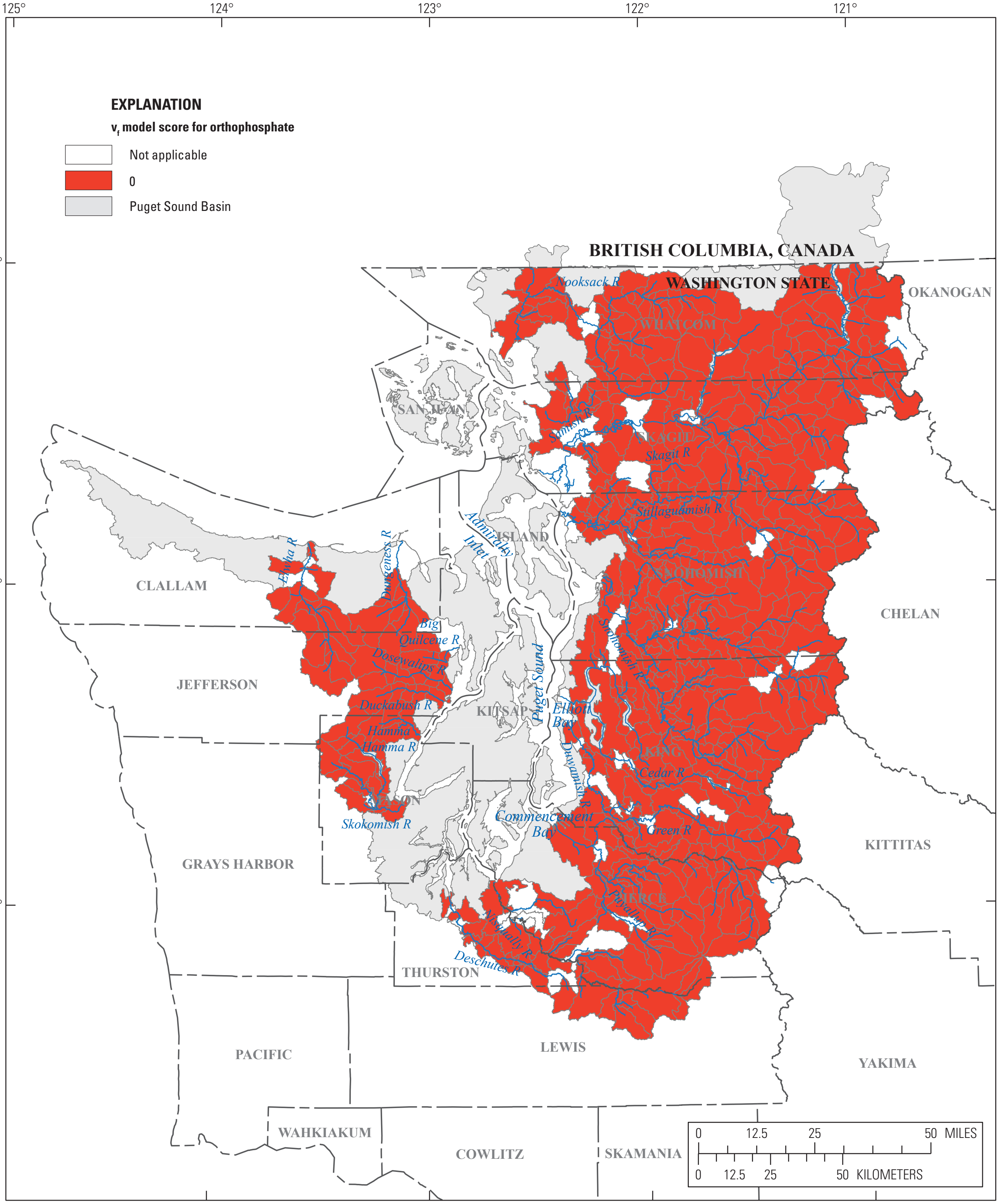


Figure B5. V_f model score for nitrate, Puget Sound Basin, Washington.



Base from U.S. Geological Survey digital data, 1:100,000
Transverse Mercator projection: NAD 1983 UTM Zone 10N

Figure B6. V_f model score for orthophosphate, Puget Sound Basin, Washington.

Publishing support provided by the U.S. Geological Survey
Science Publishing Network, Tacoma Publishing Service Center

For more information concerning the research in this report, contact the
Director, Washington Water Science Center
U.S. Geological Survey
934 Broadway, Suite 300
Tacoma, Washington 98402
http://wa.water.usgs.gov

Sheibley and others—**Nutrient Attentuation in Rivers and Streams, Puget Sound Basin, Washington**—Scientific Investigations Report 2015–5074, ver. 1.1

ISSN 2328-0328 (online)
http://dx.doi.org/10.3133/sir20155074