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Dynamics of Channel Complexity and Nitrate Retention in Upper Fanno Creek, Oregon

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Dynamics of Channel Complexity and Nitrate Retention in Upper Fanno Creek, Oregon

by

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requirements for the degree of

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ABSTRACT

This study investigates the relationship between channel complexity and nutrient spiraling along 31 reaches of an urbanized watershed in Portland, Oregon. Much research shows that urbanization has an effect on watershed hydrology and nutrient loading at the watershed scale for various sized catchments. However, the flux of nutrients over short reaches within a stream channel has been less studied because of the effort and costs associated with fieldwork and subsequent laboratory analysis of the surface water samples. In this study I measure channel complexity and uptake velocity of nitrate to determine if this relationship is indicative of a healthy, functioning stream. I take field measurements and samples to determine the complexity and uptake velocity of each reach. Using ion-selective electrodes, the fluxes of nitrate were measured within each reach; when combined with channel geometry and velocity measurements these measurements allow for the transformation of nitrate fluxes into spiraling metrics. Results show that 18 of the 31 reaches had uptake velocity. Discharge and sinuosity were positively correlated with nitrate uptake velocity. Complexity and nitrate concentration were negatively correlated with nitrate uptake velocity. Grass landcover was positively correlated with nitrate uptake velocity and negatively correlated with nitrate concentration. These results indicate that land use and channel complexity both are

related to the in-stream processing of nitrate. The implication of this study is that channel complexity is an important driver of nutrient flux in an urban watershed, and that this technique can be applied in future studies to better characterize water quality of stream channels over short reaches to entire catchments.

DEDICATION

I dedicate this thesis to Kay.

You have been and continue to be an inspiration.

ACKNOWLEDGEMENTS

I would like to acknowledge and thank my thesis advisor Martin Lafrenz and my committee members Tom Harvey and Heejun Chang for their inspiration, insight, guidance and support in research and writing of this thesis. I would like to recognize the help I received from fellow students in the Department of Geography for their assistance in the field. Finally, I would like to thank Kay, who spent countless hours assisting me in the field, organizing my thoughts, and preparing this thesis. Without your help and encouragement I would still be sloshing around somewhere in the watershed.

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GLOSSARY/LIST OF ABBREVIATIONS

COP City of Portland, Oregon.

CWS Clean Water Services – public utility company.

ISE Ion-selective electrode.

NO_2 Nitrite, form of nitrogen that is toxic, water soluble, and can convert to nitrate.

NO_3^-N Nitrate as nitrogen, reported as mg/L or ppm; nitrate (NO_3) is 4.4 x NO_3^-N.

NN Nitrate-nitrogen

NH_3^+ Ammonia, sources: fertilizer, atmospheric deposition, or nitrogen-fixation.

NH_4^+ Ammonium, sources: animal waste, effluent from sewage treatment.

PPM Parts per million.

S Slope calculation

s Sinuosity.

V_f Uptake Velocity

ε Symbol for longitudinal roughness in this paper.

X Symbol for complexity in this paper.

CHAPTER 1 – REVIEW OF HYDROGEOMORPHIC CONTROLS ON NITRATE DYNAMICS IN URBAN AREAS USING THE SPIRAL CONCEPT

What do beavers have to do with nitrate in Fanno Creek? Beavers are engineers, cutting and moving wood, creating shelter for themselves, and shifting the course of the river. I reflected on this as I explored a site where a beaver dam had been removed in upper Fanno Creek. Piles of accumulated sediment were cracked, with tiny, braided channels connecting in a chaotic picture that seemed completely at odds with the paved 4-lane highway behind me. Bulldozers and giant earth moving machines removed the remains of the dam. Trees were planted along the banks to restore the "damage" the beavers had done, complete with wire mesh at the base of the saplings, meant to send beavers the message: "We do not want your dams; we can take care of this ourselves."

This was a restoration, after all. There were concrete walls to simulate sediment trapping function of a beaver dam and anchored logs to simulate the trees beavers bring into streams. It appears that beavers have made an impression on us; consequently we have learned a bit from them. They modify streams, slow them down, whittle out detours, and provide a home and habitat for a whole ecosystem; the dissimilarity between human and beaver engineering is that the human footprint is much larger and has had unforeseen impacts.

Urban streams have a bad reputation: they flood and they are dirty. Portland is a city carved from wood and akin to the beaver; we cut down trees to engineer our habitat. As the city burgeoned and trees were felled for development, so have the streams that once flowed through forests been transformed. Over the past 30 years there have been efforts to restore streams around the country for a variety of reasons, including fish habitat, aesthetics, flood control, erosion, and water quality. However, stream restoration is expensive and contentious.

This thesis is an investigation into how linkages between biogeochemical cycles and geomorphic structure can be evaluated in order to better evaluate both stream restoration and the urban stream phenomena. In this chapter I will review existing literature to identify conceptual and empirical links between geomorphology and biogeochemical function. In Chapter Two I apply techniques to measure geomorphic and biogeochemical function, and then discuss the results and the implication for future work. This research is timely, given the increase in public interest and evolution of stream restoration work in urban areas as an ecosystem service. Therefore, this work will be of interest to managers as they have lacked a scientific basis to couple stream structure with natural ecological benefits and anthropogenic ecosystem services (Thorp *et al.*, 2010).

Hydrogeomorphology

Due to the increasing multi-disciplinary work that encompasses stream restoration, this chapter will review (1) emerging hydrogeomophic concepts including influences on solute transport, transient storage, complexity, and hot spots/hot moments; (2) the dynamics of the nitrogen cycle in an urban riparian context, including the spiraling framework, and urban spiral studies; (3) and how these concepts can be applied to river restoration, management, and monitoring.

Almost 40 years ago, Scheidegger (1973) coined the term *hydrogeomorphology,* which he defined as "the study of landforms as caused by the action of [liquid] water." To rephrase, hydrogeomorphology is "the study of the impact of hydrologic processes on the land" (DeBarry, 2004, 93). Hence, a hydrogeomorphic study would examine how land use change affects channel form. In contrast, fluvial geomorphology is the study of how river processes shape the land, for instance how a stream meander forms and evolves. This review of hydrogeomorphology is limited to mechanisms that affect solute transport, including transient storage and channel complexity.

Solute transport controls

Solutes like dissolved organic matter, salt, or chlorine, are carried with the flow of a river, and thus as the flow of the river is altered, the transport of a solute is altered. The first control on solute transport is advection (Figure 1.1, A), where solutes are picked off as they move over a rough surface. In contrast, longitudinal dispersion (Figure 1.1, B) ensues when a concentration of solute is split by the flow so that some parts of the solute are pushed to the front while others are physically forced to the back. Where the channel forces solutes into or out of the main flow is called transient storage (Figure 1.1, C). When a tributary enters a main channel, their convergent flows will alter solute concentration through the process of flowpath intersection mixing (Figure 1.1, D). These alterations are mechanical mechanisms; there is no chemical reaction although they can lead to chemical reactions.

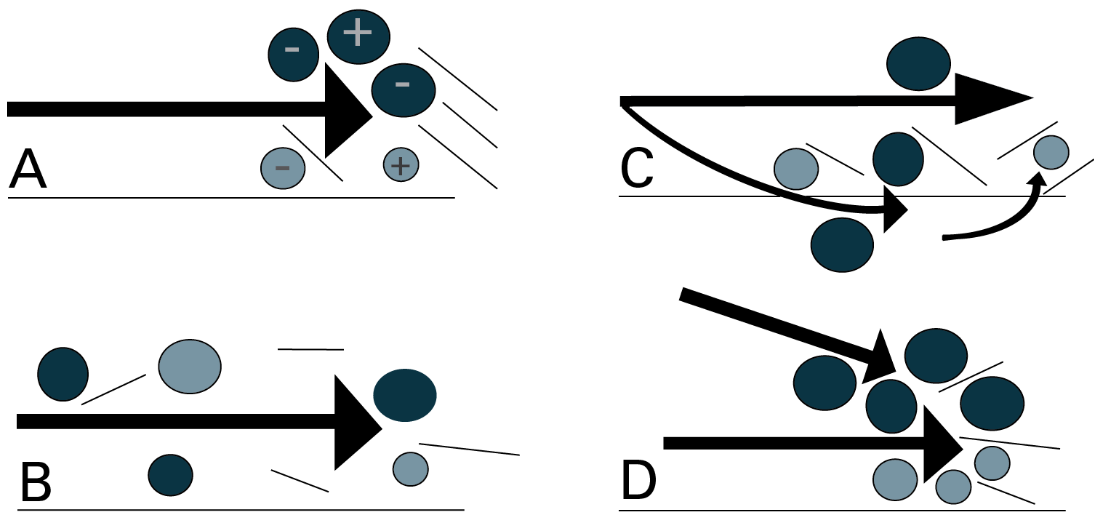

Figure 1.1. Hydrogeomorphic mechanisms that affect solute transport: advection (panel A), longitudinal dispersion (B), transient storage (C), flowpath intersection mixing (D) (adapted from Gooseff *et al*., 2008)

Transient storage

As a stream runs its course, it carries a solute load the concentration can be mechanically altered due to several processes including transient storage. Transient storage can alter the solute concentration several ways. First, transient storage can move solutes into "dead storage" (Figure 1.2, A), such as into a deeply scoured pool. The second way transient storage can alter channel solute concentration is by upwelling the solute out of the hyporheic zone (Figure 1.2, B); in contrast, the third process is downwelling of the solute into transient storage (Figure 1.2, C). Upwelling will increase the solute concentration of the stream flow whereas downwelling will decrease the solute concentration. Complex channel structures can cause many of these mechanical processes to occur over a small area.

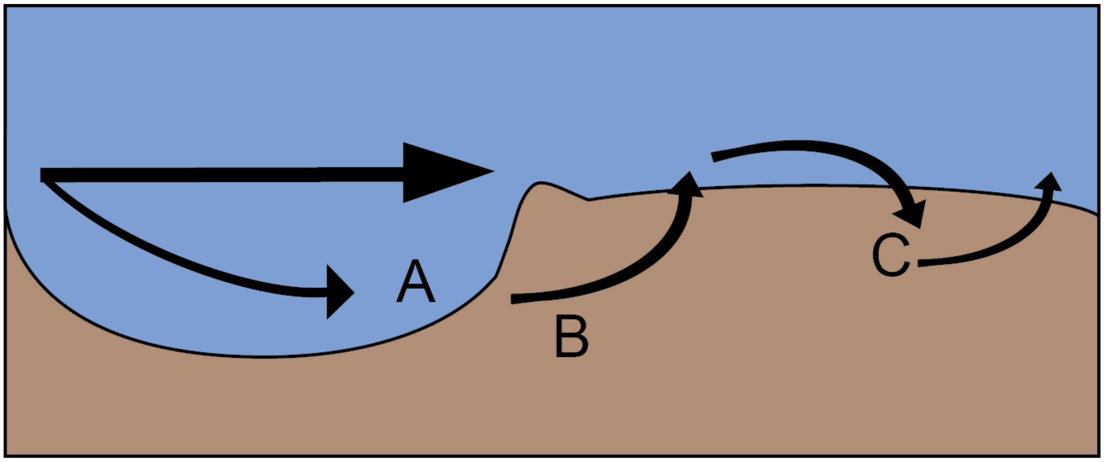

Figure 1.2. Movement of a solute through types of transient storage: (A) dead storage, (B) hyporheic upwelling, (C) downwelling

Complexity

Complexity is the spatial variation of a channel reach, including the type of vegetation in the bed and on the banks (Grimm *et al.*, 2005), the presence of large woody debris (Gregory, 2006), and the hydraulic geometry parameters of slope, sinuosity, and roughness (Gooseff *et al.*, 2007). Complexity also reflects geomorphic processes; erosion produces a scour pool while a side channel can be created from sediment deposition following a flood event. As channel complexity reflects geomorphic processes, it also structures the function of other channel-related process, including hydrology and biogeochemical cycling. Studies have shown a strong link between geomorphic complexity and transient storage, which is directly related to in-channel nitrate processing (Gooseff *et al.*, 2007; Claessens *et al.*, 2010; Baker *et al.*, 2012). Transient storage in streams and rivers includes not only hyporheic zones created by gravel bars, step-pools, and meander belts, but also "dead storage" common in urban scour pools and detention ponds. This implies that while increasing complexity will surely increase transient storage, the nature of that storage will vary depending on the nature of the complexity. For instance, large hyporheic zones with long residence times are characteristic of large meander belts connected to floodplains (Figure 1.3). Conversely, transient storage with a very short residence time is characteristic of streams with step pool morphology.

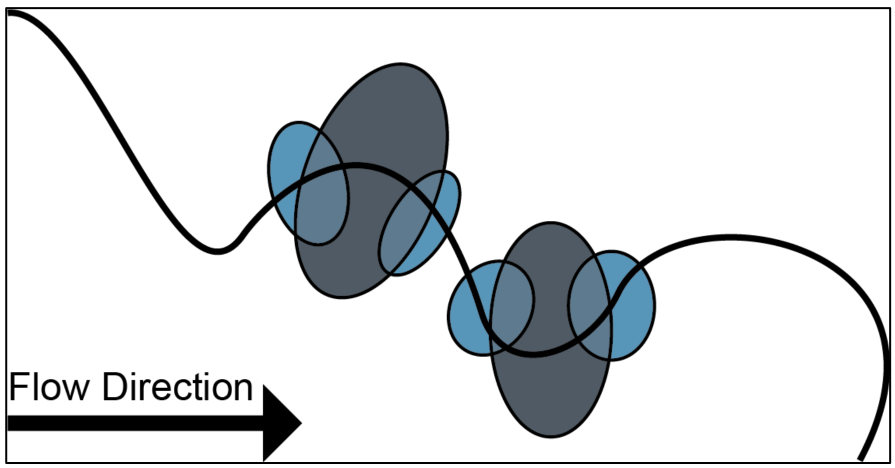

Figure 1.3. Meander belts with large hyporheic zones represented by large circles, smaller circles denote changes in size and residence time as the stream runs into and out of a meander belt

Nitrogen Cycle

Biogeochemical cycling is the transformation of elements through the atmosphere, hydrosphere, biosphere and lithosphere. Many of these transformations involve nutrients, like nitrogen, which moves in and out of reservoirs in the different spheres. Because many nutrients are coupled to other substances like water, the spatial scale at which cycles are studied ranges from a global perspective down to the flux of nitrogen in a single tree.

Nitrogen is constantly moving from the soil up through trees, into the atmosphere, and back to the soil in a process referred to as the nitrogen cycle. Trees cycle large amounts of nitrogen on a daily, even hourly basis, which is a relatively short process when compared to nitrogen cycling in a lakebed, where it may take years for nitrogen to move in and out of the substrate (Schlesinger, 1997). Each step in the cycle follows a precise path depending on conditions at multiple scales.

As nitrogen (N) is cycled, there are many pathways it can take that may change its concentration and spatial distribution. Streams and rivers are one such vector of removal and delivery. In Figure 1.4, wood releases organic-N as it decays. As the organic-N is released in situ, it is converted (mineralized) to ammonium (NH_4^+), a type of nitrogen utilized by some plants (Figure 1.4, C and E). At times there might be more NH_4^+ than is needed for plant use, or the plants at that location may not use the NH_4^+ form of nitrogen. In any case, if it is not assimilated through plant uptake as NH_4^+ (Figure 1.4, D), then nitrification can occur. Nitrification is when NH_4^+, an organic form of nitrogen, is converted to nitrate (NO_3^-), an inorganic form of nitrogen. To oxidize NH_4^+ into NO_3^-, it must first be transformed to nitrite (NO_2^-), an intermediate-inorganic form of nitrogen. Once in NO_2^- form, bacteria oxidize the NO_2^-, which creates NO_3^-, thus completing the nitrification process (Figure 1.4, B). As nitrogen is in NO_2^- and NO_3^- form, it is water-soluble and can easily be moved away from the point of origin. In nitrate form, nitrogen is easily taken up by plant roots for growth; this is another form of assimilation (Figure 1.4, D) and represents another way nitrate leaves surface waters. Alternately, in a reduced state, which is common in wetlands, slow moving channels, and lakes, conditions exist that cause denitrification to occur. Denitrifiying bacteria feed on NO_3^- to sequester organic carbon, while reduced sulfur, sulfate (SO_4^-), provides energy for bacteria to denitrify nitrate. As this process happens, nitrogen is released to the atmosphere as nitrogen gas (N_2), nitric oxide (NO), and nitrous oxide (N_2O).

If any component changes, it can disrupt the cycle, possibly throwing the ecosystem out of balance. For example, denitrification in (Figure 1.4, A) is positively reinforced by sulfur and negatively reinforced by carbon dioxide. If carbon dioxide levels decrease the process of denitrification will increase whereas when the level of sulfur lowers denitrification will decrease. There are limiting chemical and physical factors like these at all scales that determine the path and state of nitrogen. Thus, nitrate gets to stream channels through upstream contribution, overland flow, and through different processes in the nitrogen cycle that can add or remove nitrate (Correll, 1997; Schlesinger, 1997).

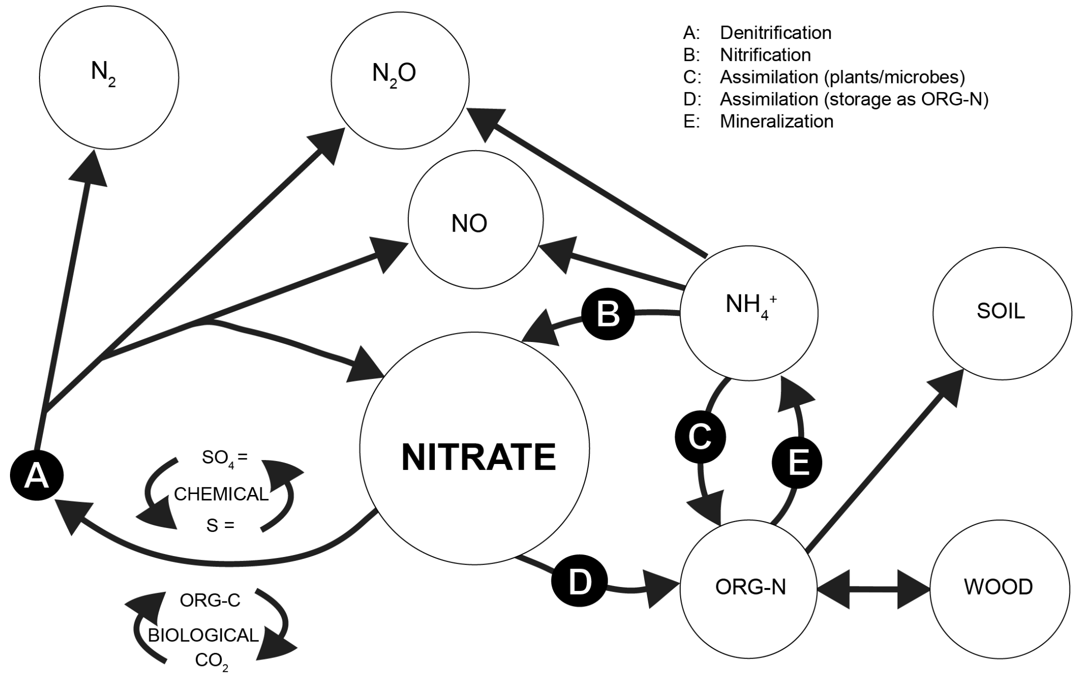

Figure 1.4. Biogeochemical processes and pathways for nitrate removal in the riparian zone (from Correll 1997).

Spiraling Theory

As water moves through a channel, dissolved nitrogen is transported in and out of the channel substrate and transformed through various forms. Thus, the same processes of transformation that happen in a terrestrial nitrogen cycle also happen in a stream channel, but the dynamics are different due to the fact that the stream is in continual motion. This process of nutrient cycling in a stream is termed spiraling; Newbold *et al*. (1983) developed a nutrient spiral framework to explain these nutrient processes in stream channels.

Spiraling is a function of three components, including uptake velocity, v_f, spiral length, s_w, and areal uptake, U. Uptake velocity, v_f, is the rate that a constituent (e.g. nitrate) moves from the stream through the water column into the benthos and transient storage, which includes "dead storage" (Figure 1.5, B). Spiral length, s_w, is the distance a constituent (e.g. nitrate) travels along a channel before completing the spiral process and moving back into the channel substrate (Figure 1.5, A). Areal uptake, U, is the amount of uptake by vegetation (Figure 1.5, C). The relationship between these has traditionally been interpreted as a first-order, linear process (Stream Solute Workshop, 1990).

Initially, spiraling studies were conducted using isotopic tracers (e.g. Newbold *et al.*, 1983) to track movement of a particular constituent. Later, studies used additions of nitrate to measure the amount of uptake in a channel. Further methods were adopted that used conservative tracers to measure flow through transient storage combined with isotopic tracers to measure a constituents movement relative to the flowpath; other studies used conservative tracers with additions of nitrate to simulate and measure response to natural conditions (Teissier *et al.*, 2002; Grimm *et al.*, 2005; Wollheim *et al.*, 2005; Wollheim *et al.*, 2008a). In sum, spiral studies have been intensive over small areas with mixed results.

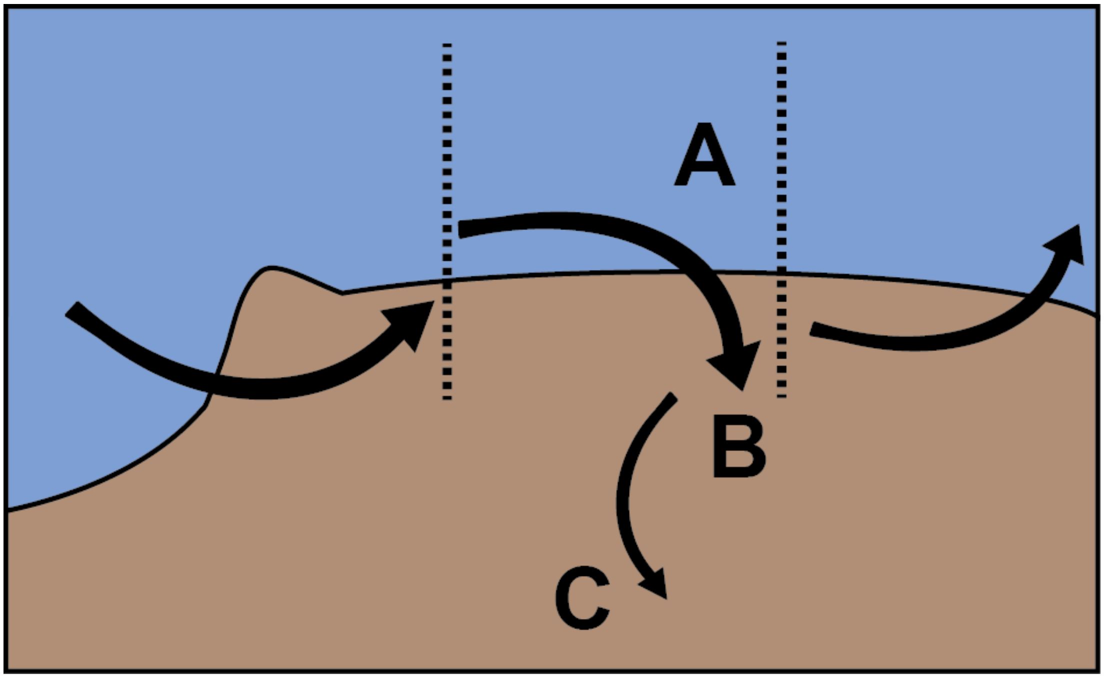

Figure 1.5. As a spiral moves downstream (left to right) the primary components are the spiral length, s_w, (A, above) is the length that an element travels until it has completely exited the channel flow, the uptake velocity, v_f, (B, above), the rate an element moves out of the channel flow, and areal uptake, U (C, above) the uptake of elements by vegetation or bacteria (Stream Solute Workshop, 1990)

Recent developments in spiral theory include refutation of using a first-order kinematic process (i.e. linear, steady state) to represent the transport and transformation of nitrate (Claessens *et al.*, 2010). The alternative to first-order process models involves using higher order functions like Michaelis-Menten, an enzyme function that is non-linear and will reach zero (no uptake) as saturation increases. While the Michealis-Menten function has been widely used to model spiral dynamics, it is not optimal for every circumstance, and some studies indicate that it may not be representative at the watershed scale (Tank *et al.*, 2008; Baker *et al.*, 2012; Claessens *et al.*, 2009). Difficulty arises because of the complexity stream networks show at different scales. For example, local transient storage may vary from reach to reach in the same pattern (e.g. Figure 1.2), which could have an effect on the entire watershed. However, if the watershed was saturated with nitrate, transient storage would have little effect on the export of nitrate at the watershed scale.

Noting that riparian areas spatially vary and recognizing that many biogeochemical processes are non-stationary–that is to say they vary from location to location–McClain *et al*. (2003) introduced the concept of hot spots and hot moments in riparian areas. Hot spots are zones of increased activity and hot moments are temporal zones of increased activity. An activity that may be enhanced could be a biogeochemical process, like denitrification, in a hyporheic zone that has flow and delivery of nitrate regulated by the river (Groffman *et al.*, 2005; Vidon *et al.*, 2010). In this instance a hydrogeomorphic process, the river flow, and biogeochemical process, denitrification, intersect to create a hot spot or hot moment of denitrification. Hot moments are of particular interest because it is theorized these occur in response to high intensity events, like a sediment pulse in a flood event (Trimble, 2010). Thus, it is important not only to determine where processes intersect in 2-d space but also in 3-d space-time as a network may experience space-time cascades that will change over time. The implication of hot moments for restoration and rehabilitation efforts is that the entire system must be considered and that one location may not solve a problem in the same way through space-time. New advances in distributed stream modeling can include hot moments, which may represent a period of time when more movement of solutes takes place (Riml and Wörman, 2011), but this is certianly an emerging field with little empirical data.

Urban Spiraling Studies

Several studies have investigated uptake velocity in urban areas with most research having been done in the Eastern United States (e.g. Boston, Wollheim *et al.*, 2005; New York City, Newbold *et al.*, 2006; Baltimore *Long-Term Ecological Research* [LTER] project, Claessens *et al.*, 2009) along with a study in the Phoenix LTER (Grimm *et al*., 2005) and one in Colorado, (Baker *et al.*, 2012). Figure 1.6 shows a comparison of urban spiral studies to other non-urban studies over the past 30 years. An urban spiral study done in the Baltimore LTER had low uptake velocity while the Arizona LTER study had moderate uptake velocity when compared with the non-urban studies reviewed by Ensign and Doyle (2006) and Tank *et al*. (2008). The Baltimore study revealed higher concentrations of NO_3^- than the Arizona study, which showed a wider range of NO_3^- concentrations. These results do not appear to demonstrate any pattern that differentiates urban from non-urban streams with respect to nutrient spiraling; however, much more data is needed to make any definitive statements. At the present I have found only one study documenting nitrate retention or uptake in an urban stream in the Pacific Northwest. Sonoda's (2002) dissertation includes data on nitrogen dynamics in urban watersheds; however, the focus is on the effect of land use on nutrient concentration and does not assess in-stream dynamics. Two other studies have also tangentially looked at the movement of nitrogen, phosphorus, and dissolved organic matter through urban watersheds (Sonoda *et al.*, 2001; Hook and Yeakley, 2005) but again, do not measure in-stream processes.

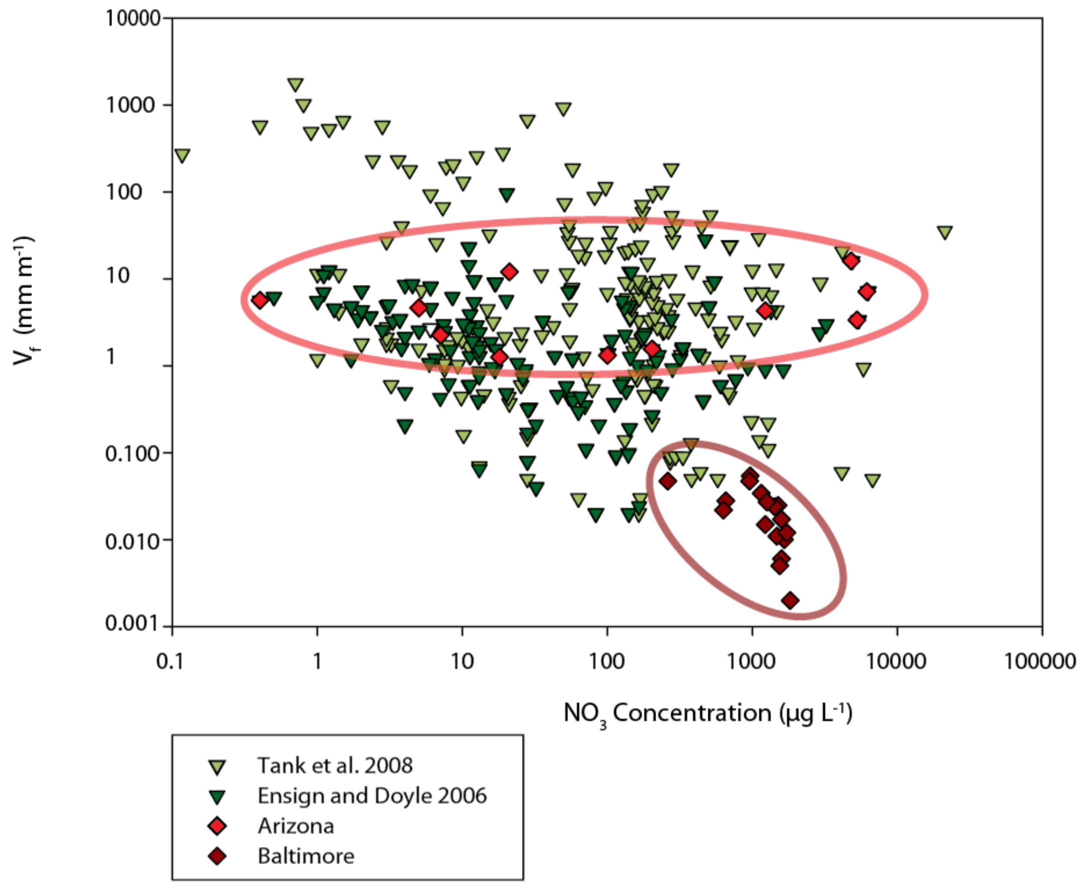

Figure 1.6. Comparison of spiral studies which looked at nitrate uptake velocity, based on databases in Ensign and Doyle (2006) and Tank *et al*. (2008) that are both compilations of spiral studies over the past 30 years; raw data from urban spiral studies in Arizona (Grimm *et al*., 2005) and Baltimore (Claessens *et al*., 2009) both have generally lower uptake velocity rates than non-urban studies

Conclusions

Implications of Studies

The spiraling phenomenon shows promise as a river restoration technique because it is a framework that readily integrates with multiple disciplines, such as riparian ecology and fluvial geomorphology. Further, uptake velocity has been shown to be a parameter sensitive to urbanization (Newbold *et al*., 2006), and might also be a good monitoring tool for watershed management.

Management of streams and watersheds can use the spiral framework as a function of channel complexity to make more informed decisions when evaluating sites for potential restoration, as well as the efficacy of existing restoration with respect to flood control and water quality restoration goals. Recognition that hot spots and hot moments are naturally and anthropogenically created through biogeochemical and hydrogeomorphic processes (e.g. denitrification and transient storage) can help design hot spots for removal under different conditions (e.g. stormwater hot spots and baseflow hot spots). Complexity is directly related to transient storage. The residence time of a solute in transient storage is mediated in part by complexity; moreover, longer residence time in transient storage increases the potential for denitrification or assimilation of nitrate.

Future Work and Research Gaps

Integrated studies that combine biogeochemical and hydrogeomorphic principles have been limited to date and heavily biased towards the Eastern United States and LTERs. Studies have already shown the fundamental processes at work in the Pacific Northwest watersheds are different than those in the the Eastern United States (Schaefer *et al.*, 2009); hence, there still exists a need to evaluate urban watersheds in the Western United States both from a physical hydrogeomorphic perspective and an ecological biogeochemical perspective. The spiral framework provides an ideal platform to conduct integrated studies that combine biogeochemical and hydrogeomorphic principles, given its accessibility by multiple disciplines, like hydrology, ecology, and geomorphology. Meanwhile, advances in numeric modeling may facilitate this research by making the spiral concept more easily modeled by managers using publicly available software (e.g. United States Geological Service [USGS] *One-Dimensional Transport with Inflow and Storage* [OTIS] available at: http://water.usgs.gov/software/OTIS/).

CHAPTER 2 – INVESTIGATION OF REACH-SCALE STRUCTURAL INFLUENCE ON NITRATE DYNAMICS, FANNO CREEK WATERSHED

Introduction

In 1993, the Oregon Department of Environmental Quality listed Fanno Creek on the US Environmental Protection Agency (USEPA) 303(d) list for temperature, phosphorus, and ammonia (City of Portland, 2005). Years of tree clearing for house building, industrial dumping, and management that consisted of pouring concrete on the banks to prevent erosion scarred the landscape. In response, the City of Portland began river restoration efforts and investment, including the creation of watershed councils, friends of rivers groups, and dozens of capital improvement projects with the goal of improving water quality. Because of this effort, and others like it on a national scale, the river restoration industry has grown into a billion-dollar-a-year business (Kondolf, 2006). As cities develop, stream miles affected by urbanization will only increase, which, in turn, will require additional remediation under the current management model. Consequently, it will become ever more important for us to improve the management, protection, and revitalization of our urban streams and rivers in such a way that increases water quality, allows for flood control, maintains suitable habitat for wildlife, and provides recreation opportunities in a cost effective manner.

Under the auspices of river restoration, this paper is an examination of how channel complexity, the landscape a river flows through, cleans water by "turning over" nutrients in a spiral. This research is specific to the effects of channel complexity on nitrate spiraling in Fanno Creek, Oregon. However, the results may be broadly applicable in other urban streams. I propose that a better understanding of channel-forced nutrient cycling in Fanno Creek can increase our understanding of the efficiency of nutrient spiraling in other urban streams. In short, channel complexity could be a new approach to river restoration and watershed management that mitigates water quality issues with a minimal amount of stream engineering.

Approaches to river restoration in the United States since 1990 have generally followed natural channel design (NCD), a step-by-step method introduced by Rosgen (1994). Typical restorations would first include a characterization of the reach to be restored based on its morphology, including any meanders, pools, riffles, and bed material as well as vegetation. Then, based on the reach classification, a template would be chosen as a restoration guide that bests approximates a natural condition. Common features of this approach include anchored large woody debris, rip-wrap or other of bank stabilization features, and vegetation plantings (Wohl *et al.*, 2005). A main criticism of NCD is that the restorations are rarely monitored; hence, the success of this approach has been the topic of many debates (so-called 'Rosgen wars') (Lave, 2009, 2012).

Recently, there has been a large body of work calling for trans-disciplinary research on river restoration (Bukaveckas, 2007; Dufour and Piégay, 2009; Bennett *et al.*, 2011; Nestler *et al.*, 2011; Violin *et al.*, 2011). The concept is that teams of scientists should examine the local hydrology, geomorphology, and ecology of a stream and make recommendations for restoration strategies based on desired outcomes. One such outcome is to restore in-channel biogeochemical processing (i.e. nitrate removal). A trans-disciplinary approach recognizing the dynamics and drivers of the watershed, creates restoration possibilities that are more than landform constructions.

In all ecosystems, biogeochemical processes add or remove substances to varying degrees through a series of cycles. Central to this study is the nitrogen cycle. In riparian areas, frequent disturbances caused by biotic (animal herbivary) and abiotic (e.g. flooding, fluctuating water tables), have resulted in species that have adapted to these unstable conditions. Nitrogen fixers convert atmospheric nitrogen to ammonia (NH_3^+), which is a form of nitrogen other plants can use to grow. In urban areas, sources of nitrogen include industry (atmospheric N_2), automobile exhaust, landscaping (fertilizers: nitrate, NO_3^-, nitrite, NO_2, and ammonia, NH_3^+), and waste (including septic systems, combined sewer-stormwater overflows, and wastewater treatment). Impervious surfaces also enhance urban nitrogen contribution. As dry deposition accumulates in urban areas, nitrate is mobilized by water in runoff following precipitation and by human activity, such as watering of fertilized lawns, and delivered to riparian areas.

Nitrification is the process that creates nitrate under aerobic conditions. The process is affected by temperature (the bacteria *nitrobacter* is half as productive at temperatures less than 20ºC) and pH (nitrifying bacteria need a pH of 6.8 to 7.3) (Keeney, 1973). Increases in cloud cover can lower the primary production and vegetative uptake demand, which in turn lowers the amount of nitrate removed from a stream reach by vegetation or bacteria (Keeney, 1973). Hydrology also has an effect on nitrate flux; as surface and groundwater interact, reaches that are “gaining” or “losing” water from local aquifers may also be “gaining” or “losing” nitrate. Climate change has an effect, as urban wetlands have lower productivity due to lower water tables and fewer anaerobic areas, which are needed to promote denitrification or immobilization (Ehrenfeld, 2000; Gaston *et al.*, 2010; Filoso and Palmer, 2011). Dissolved oxygen levels affect channel nitrate transformation when low levels create anoxic conditions that promote denitrification leading to the algal blooms that are indicative of excess nitrogen.

In 1975, Webster coined the term spiraling, which refers to the nitrogen cycle acting within a stream channel. As water moves through a channel the concentration of dissolved nitrogen (including nitrate) changes creating different fluxes along a reach of a stream channel. Newbold *et al*. (1983) expanded this idea and explained how the system worked conceptually. As water flows along a channel, inputs of nitrate are added to the stream from hillslopes. However, rather than seeing increases in nitrate concentration Newbold and his colleagues found that as the stream flowed, the nitrate concentration decreased. In fact, when they added nitrate its concentration decreased a little more indicating *uptake* of nitrogen over a stream reach.

Newbold *et al*. (1983) tested the rate of spiraling by addition of nitrate and through use of a tracer. The addition method involved loading a concentration of nitrate at a particular point on the river and then taking regular measurements downstream to determine the flux of nitrate along a reach (Fisher *et al.*, 2004). The alternate method involved depositing a radioactive tracer into the river at a certain point and following it, while noting its path, speed, and exit point to determine how much had been taken up by vegetation (Grimm *et al.*, 2005).

Initially these experiments seemed to indicate that despite any attempt to overload the stream, nitrate concentration ultimately emerged downstream lower than it originated. They hypothesized that trees and other vegetation were absorbing the nitrate. Subsequent research discovered that only a fraction of the nitrate actually went to the vegetation (Wollheim *et al.*, 2008a; Wollheim *et al.*, 2008b). Thus, there must have been another mechanism of uptake and storage.

Further experiments conducted by Newbold revealed that what actually occurred was more complicated. Isotopic studies showed that channels with relatively more gravel, curves, and variations contained larger amount of transient storage potential for nitrogen. Transient storage allowed pathways for nitrogen including the slow release of accumulated nitrate to vegetation, the release into the atmosphere as gas, the release as a solute into the stream, or continued sequestration in the stream bed (Golterman, 2004). These processes created spirals both over space and time in the channel.

To summarize: several factors influence the degree to which nitrate concentrations fluctuate in a stream channel, including 1) the inflow of nitrate from sources upstream, laterally, or through the hyporheos); 2) the type and distribution of riparian vegetation; and 3) the fluvial geomorphology of a channel. Generally, urban areas generate a great more nitrate than non-urban areas (except agricultural areas) which can lead to high concentrations of nitrate and subsequent water quality issues.

Concentrations of nitrate in urban areas can vary, and the cause of this variance is unclear due to complex dynamics that drive nitrogen cycles.

Complexity and Retention Dynamics

Atmospheric N, which can change due to land use, is deposited via precipitation. Vegetation mediates the atmospheric N delivered to surface waters via through fall. As precipitation moves through the tree canopy, water can be intercepted and chemistry altered as it moves to surface water. The density and type of stormwater infrastructure system in place mediates the path of dissolved N in surface waters. The surface-groundwater hydrology then mediates the extent external influences will modify the surface water chemistry.

Once N has entered a reservoir, there are several mechanisms for removal. These processes result in a net export of N, or a general decline in N at the outlet of the reservoir. Whether or not a reach exports or retains N is dependent on several processes, including transient storage, vegetative uptake demand, and anoxic conditions in the substrate. Anoxic conditions allow anaerobic bacteria to persist that can denitrify the NO_3^-, thus removing it. Vegetation can uptake pools of NO_3^- for net primary production; however, some riparian vegetation may also be sources of NH_4^+ or NO_3^- (e.g. *Alnus rubra*). Transient storage can serve as a storage mechanism where NO_3^- is retained for vegetative uptake, leached into the substrate and immobilized, or weakly absorbed by organic matter.

The structure of a reach can thus be influenced by land use through modification of storm water runoff, but also in the type of vegetation present. In reaches with high levels of nutrient additions, invasive colonizers (e.g. *Rubus armeniacus*) are prolific, and they can change the nutrient dynamics through demand for Nitrogen and also by lowering base flow by increased evapotranspiration rates (ET). Given that land use also modifies ET rates, structure can enhance or blunt the effects of them by forging complex pathways where multiple processes may interact synergistically to reduce or increase NO_3^- levels.

Objectives

The aim of this study is to understand the dynamics between channel complexity and nitrate concentration in an urban watershed (Fanno Creek). In order to evaluate this process, I formulated two research questions with associated hypotheses.

(1) What is the uptake velocity and spiraling nature of nitrate in a 4000 hectare urban watershed?

(2) What is the relationship between channel complexity and nitrate uptake velocity in an urban watershed with spatially varying land use and land cover?

Null Hypotheses

I hypothesized that:

H_0: Nitrate spiraling rates are not a significant response to complexity.

H_0: Complexity and land cover are equal determinants of spiral rates.

Assumptions

I make several assumptions in order to test my hypotheses. First, some local controls on channel complexity at the reach scale are fixed including the presence of bedrock, alluvium, tributaries and the role of tectonics. Second, other local controls at the reach scale are variable including weather, vegetation, and human influences (Piégay and Schumm, 2003). Third, land use and vegetation are the principle abiotic and biotic controls on the amount of nitrate delivered to a particular reach (Meyer *et al.*, 2005). Fourth, the sites selected by Clean Water Services (CWS) (2000) for rapid stream assessment (RSAT) represent the range of reaches found in the Upper Fanno Creek drainages. Finally, the first-order kinetics used by the Stream Solute Workshop (SSW, 1990) to calculate uptake velocity, v_f, can be used with field data to estimate uptake velocity, which allows me to infer the possible uptake rate without employing costly isotopes or additives (SSW, 1990).

Study Area

Fanno Creek, Portland, Oregon, is highly urbanized with at least 84 percent of its drainage area developed according to recent studies on the region (Duh *et al.*, 2008; Chang *et al.*, 2010). The catchment drains multiple suburbs and a network of stormwater and sediment detention ponds one of the most urbanized catchments in the entire Portland metropolitan area (Jung *et al.*, 2011). Fanno Creek has a complex land use legacy. Originally forested, 260 hectares of the watershed were first settled and farmed by the Fanno family in 1847; they grew onions. In 1880, John P. Hoffman cleared land near

Vermont Creek for a dairy farm. That legacy remains today as the Alpenrose dairy headquarters (both industrial and commercial enterprises) are located in the middle of the watershed. Streetcar lines nurtured suburbs that sprang up along its banks as the Portland metro area grew (City of Portland, 1994). Expansion after World War II ignited industry and commerce in the basin. Construction of massive highway projects and wide streets, buried sections of the catchment, which became covered, diverted, and polluted. As such, Fanno Creek has higher degradation of water quality, including nitrate, compared to other streams in the Portland metro area (Pratt and Chang, 2012).

Although nitrate is a nutrient essential for life, in excess quantities it is harmful to the health of humans, plants and animals. Blue baby syndrome, though rare, is attributed to nitrate exposure and occurs when babies cannot process excess nitrate (Tan, 2009). A more common issue is that nitrate leads to the eutrophication of stream pools causing fish mortality. Lower dissolved oxygen caused by algal blooms causes asphyxiation as the fish take in nitrate-nitrogen rather than oxygen (Pinay *et al.*, 2009). Nitrate has also been linked to cancer in adults (Boffetta and Nyberg, 2003); however, there is still debate about the extent and types of cancer that nitrate is associated with (Barrett *et al.*, 1998). Nevertheless, excess nitrate does lead to the poisoning of riparian vegetation (Naiman *et al.*, 2005). Surface nitrate uptake in plants may result in the poisoning other fauna in riparian areas (e.g. beavers, nutria, deer, cattle). Briefly, excessive nitrate levels can lead to plant mortalities that culminate in a stream (e.g. large woody debris) and lead to additional export of nitrate-nitrogen.

The degradation and classification of Fanno Creek as polluted water in by Oregon for temperature, dissolved oxygen, bacteria, and phosphorus, made it subject to many regulations and monitoring (City of Portland, 2005). Much of the focus has been on how the pollution is produced and radiates throughout the catchment. Roads, buildings and construction lead to an increase in impervious surface; water does not move into the ground, but rather, it flows straight into the river (Dunne and Leopold, 1978). More urban structures and increased population amplify the nitrate concentrations in the water. Stander and Ehrenfeld (2010) reviewed how riparian function is altered in urban areas, including the concentrated inputs of pollutants. For example, a wastewater treatment plant that discharges treated water into the river actually supplies pollution at various times in Fanno Creek (Smith and Ory, 2005). Sewage and stormwater systems swell during the winter as drains and sewers reach their capacity. Storm drains overflow into the river inundating the stream with untreated wastewater.

Experiments and studies have explored applying green technology to remove pollution from the watershed (Wells *et al.*, 2008). These experiments include constructed wetlands, biofiltration ponds, detention ponds, bioswales, green streets and green roofs all built in an effort to mimic the natural processes that have been thwarted by the construction of our cities. The purpose here is not to evaluate other green technologies, but rather to examine an untapped resource in restoration thought and design: the use of the channel itself for improving water quality.

Fanno Creek, (Figure 2.1) is drained primarily by first to third order streams. The basin is elliptical, flowing southwest from northeast, through multiple suburbs and a network of stormwater and sediment detention ponds. The main stem and several tributaries, including Sylvan Creek, Vermont Creek, Woods Creek and Ash Creek, were included in this study. Fanno Creek and tributaries have different hydrology than streams in other watersheds in the Portland metro area, due to differences in land use and geology (Chang, 2007).

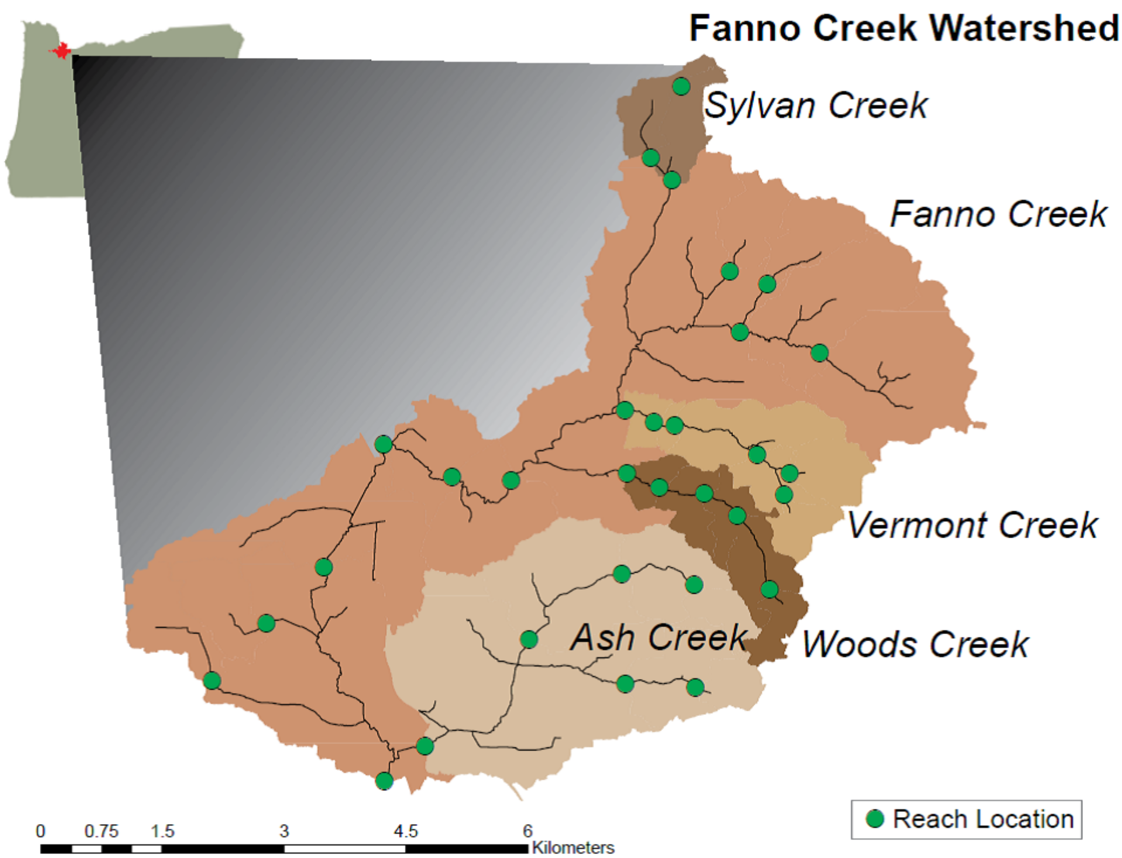

Figure 2.1. Study area, 4400 ha, and study reach locations (from Clean Water Services, 2011).

The geology of the basin is Columbia River Basalt, which is exposed in many of the northern parts of the watershed. The soils are Alfisols, Mollisols, Entisols and Inceptisols including mostly fragixeralfs, -ents, -ols, and –epts. The entire watershed has naturally high levels of phosphorus due to the presence of Apatite in the local basalt, which weathers to labile phosphorus. A fragipan, mostly of silt, extends through the entire watershed (City of Portland, 1994). The elevation of the watershed ranges from 46 to 250 meters. Most land is developed in the lower reaches, primarily residential, with the only significant forested land residing on the steep slopes in the northeast (Figure 2.2).

The Fanno Creek watershed has undergone several phases of development. In 1890, a community of several thousand households, Garden Home, was one of the first neighborhoods in the watershed to incorporate planned development. Linked to Portland via road and streetcar, areas of the watershed experienced building booms post World War II, in the 1970s, and again in the 1990s-2000s (Metro, 2012).

The current land cover is 84% developed and comprised mainly of single family residential property. However, there are pockets of commercial, mixed, and multi-family housing throughout the watershed and within some riparian areas. Thus, Fanno Creek is homogenous at a watershed scale, but at the subwatershed scale it is heterogeneous.

Water quality monitoring in Fanno Creek watershed has been temporally extensive, with monthly samples taken in 3-5 locations by Portland Bureau of Environmental Services since 1990; but these locations are spatially limited and do not include several tributaries. This can be problematic, for example, Vermont Creek was classified as a degraded stream due to severe sediment loads, yet there is no regular monitoring of Vermont Creek, although some reaches have been restored for water quality purposes (City of Portland 2005). There are little or no water quality data for

Woods Creek and Sylvan Creek which drain large parts of the watershed. Thus there is a need for sampling throughout the watershed to establish current conditions and identify suitable locations for future monitoring and restoration work.

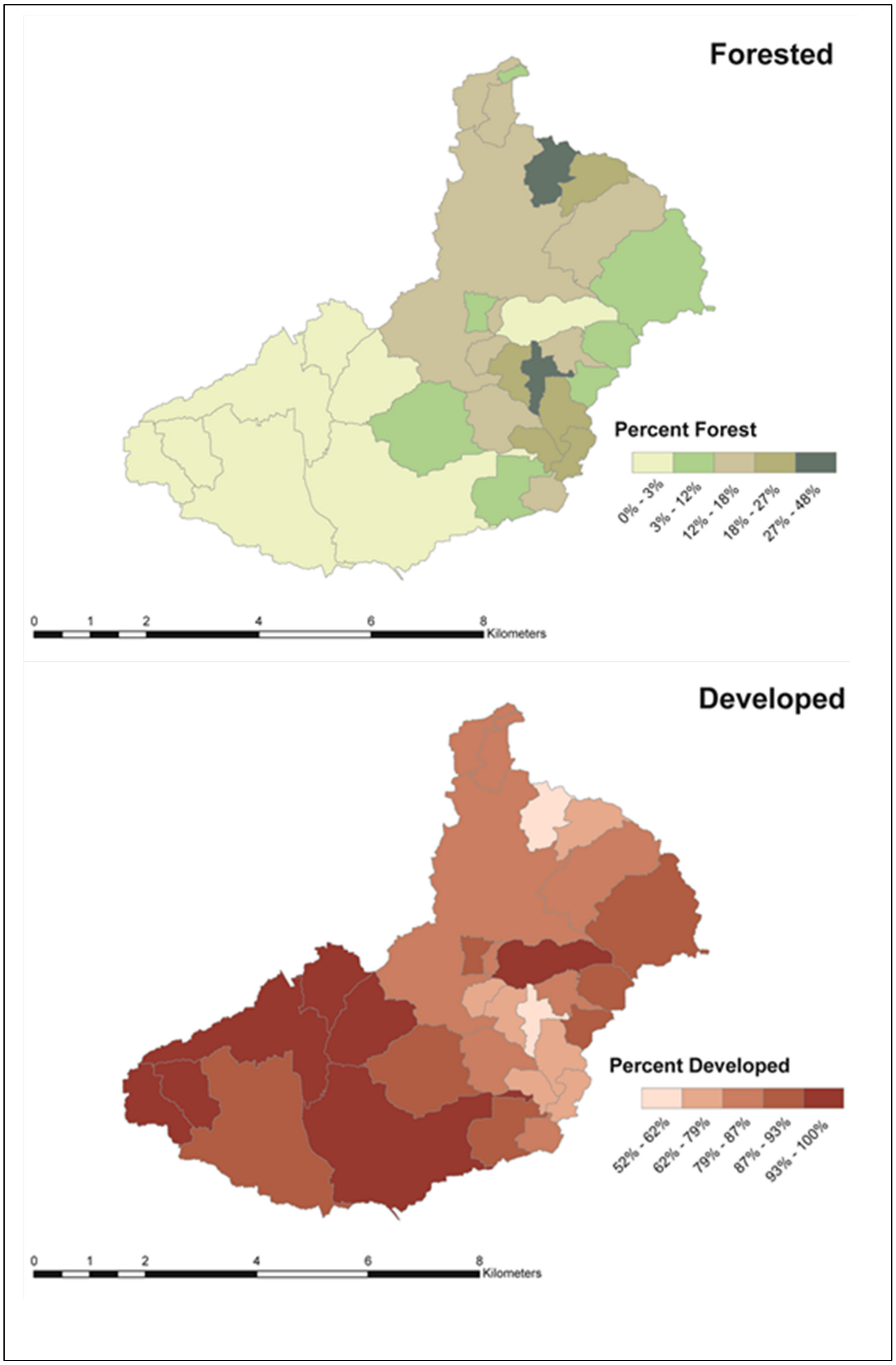

Figure 2.2. Study area land cover, percentages, forested (top); developed (bottom) (created from United States Geological Survey, 2011).

Methods

Site Selection

Each sample site, a stream reach, was selected using a GIS dataset from a watershed assessment conducted in 2000 by CWS (2000). Using the CWS survey points (N=51), I was able to enhance an already rich dataset that can be utilized for long term monitoring of change within the watershed (Figure 2.3). I chose a subset of 31 reaches within the upper Fanno Creek watershed that were publicly accessible, had perennial flow, were free of pipes within the site, were not completely channelized, and did not have cemented beds. These criteria were imperative in order to control, to the extent possible, any external contributions of nitrate within the study reach.

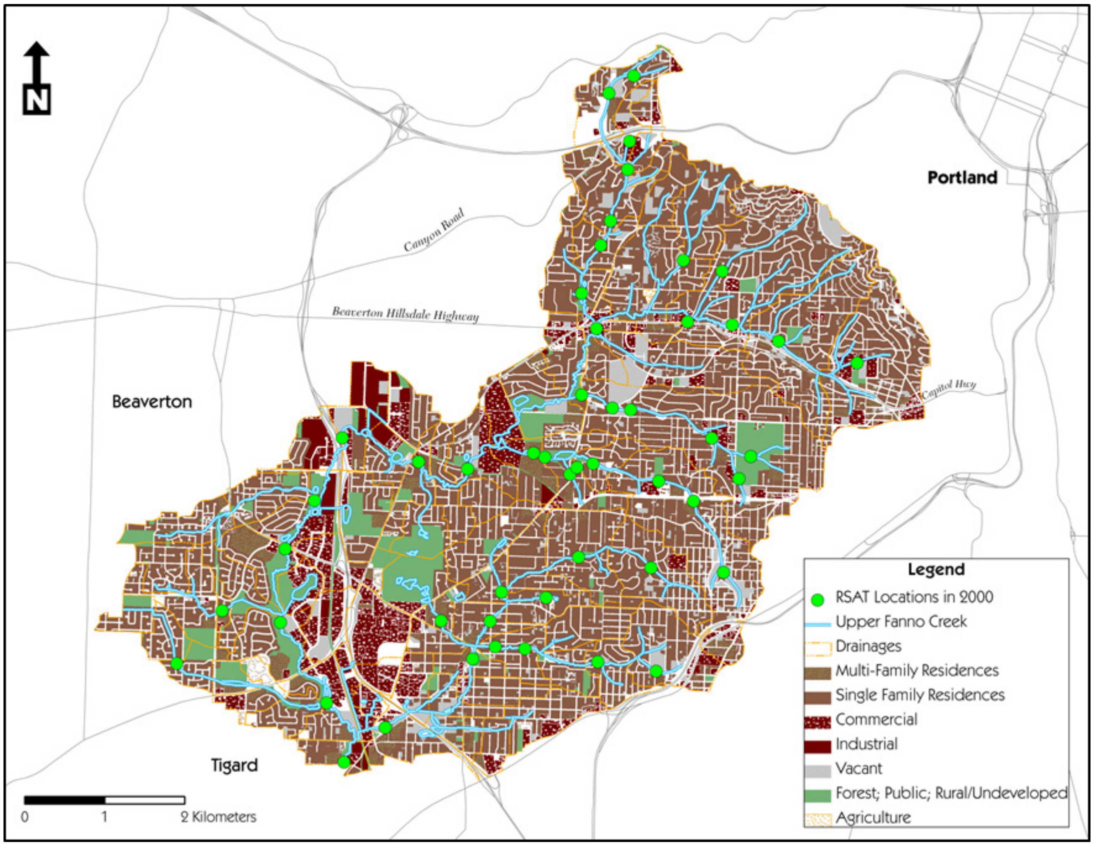

Figure 2.3. Study area by land use types (created from Metro, 2012)

Sampling and Laboratory Analysis

At each site, a reach was selected that consisted of a single geomorphic unit, such as one meander bend or one pool-riffle sequence. The reach was divided into sections with three stations: upstream (US), midstream (MS), and downstream (DS). This separation allowed me to characterize the inflow (US), outflow (DS), and any change with the reach (MS).

At each station, I recorded the location using a Trimble GeoXT with hurricane antenna capable of sub-meter accuracy. Next, I collected surface water samples using 250 mL Nalgene bottles (Fordyce *et al.*, 2005). The samples were collected by filling the bottle three times with water flowing downstream and sealing the bottle while submerged. I then surveyed the hydraulic geometry, including wetted width and depth of the channel, and measured stream velocity with a Marsh-McBurney electromagnetic meter (Harrelson *et al.*, 1994), which allowed me to accurately measure velocity in small, shallow reaches. Finally, I photographed and noted vegetation and land use at the site.

Precipitation, if present was noted, and sites that had received precipitation within 24 hours were determined by checking the Portland *hydrological data retrieval and alarm* (HYDRA) network, which had two stations located in the upper and middle portions of the watershed (Figure 2.4). All sites were surveyed and sampled between July and November, 2011—the dry to early wet season. Samples were taken during this time in order to capture the hydrologic and biogeochemical variability associated with summer-low and fall-high flows. Half of the reaches were surveyed within 24 hours of a rain event to capture hydrologic variability.

Water samples were transported in a cooler and stored at 4°C. To prevent degradation, samples brought in from the field were treated with Boric Acid (2.5 mg). I tested the samples for NO_3^- within 3 weeks in our Geography Laboratory using standard method 4500- NO_3-D (American Water Works Association, 2005). To reduce bias, samples were numbered and tested blindly. There were no replicates taken in the field. All samples were allowed to come to lab room temperature (25° C, ±1° C) prior to testing. Using an autopipette (Termo Finnpipette F1), 10 mL of sample was transferred into a 50 mL glass beaker, and then 10 mL of Thermo Nitrate Interference Suppressor Solution (NISS) was added to the sample. Next, samples were mixed for 1 min using a stir-bar; the nitrate ion-selective electrode (ISE; Thermo 9707BNWP) was then lowered into the sample-NISS mixture. The ISE was connected to a Thermo Dual-Star pH/ISE meter, which logged the nitrate concentration (mg L^{-1}) for every sample.

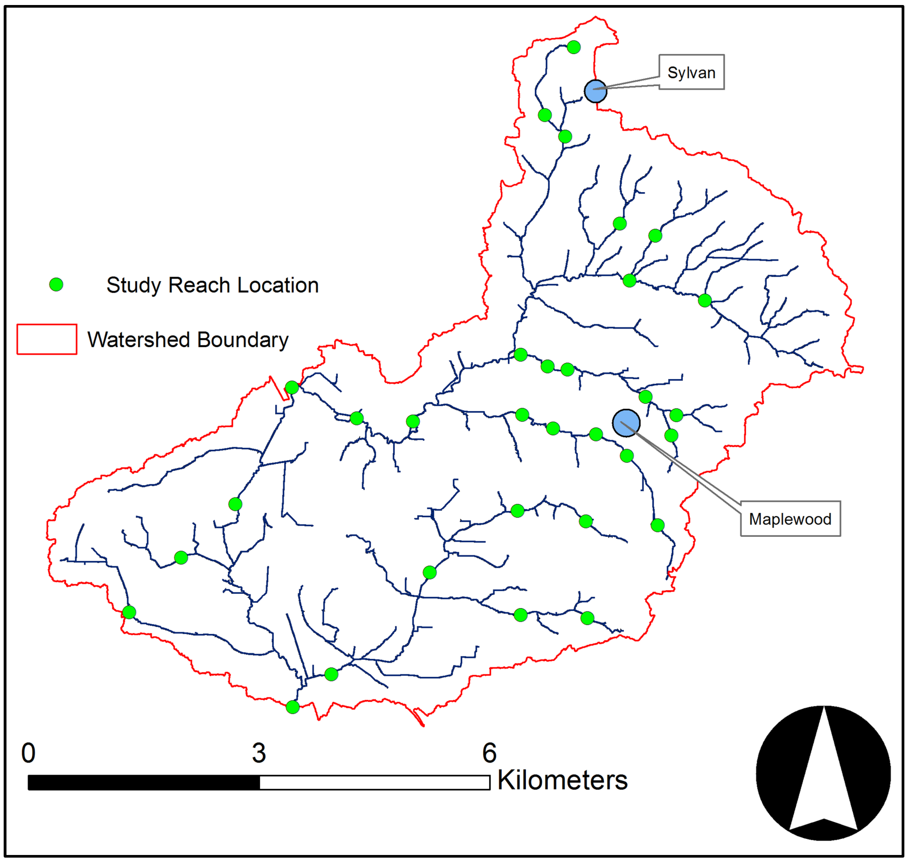

Figure 2.4. Location of Portland hydrological data retrieval and alarm (HYDRA) rainfall gages used in this study (City of Portland Bureau of Environmental Services, 2011).

Relative Contributing Areas and Complexity

I used Trimble GPS Pathfinder software to differentially correct the locations of stations. Once corrected, the points were imported to ArcGIS 10 and snapped to a stream grid. The stream grid was verified against 6-inch orthophotos (Metro, 2008). It was important for these grids to be accurate given that I would use this grid to calculate channel complexity.

The complexity index, X is the product of slope, S; sinuosity, s; and longitudinal roughness, ε (from Gooseff *et al.*, 2007)). Slope was calculated using the change in distance and elevation from the US to DS stations. The elevation of each station was determined using 3-foot LiDAR from City of Portland (COP, 2007) with 3D-Analyst in ArcGIS 10. Sinuosity, slope, and longitudinal roughness (Table 2.1) were computed using 3D-Analyst in ArcGIS 10. Finally, slope, sinuosity, and longitudinal roughness were combined to create a metric of geomorphic complexity (Table 2.1).

Table 2.1. Symbols and formulas for determining complexity used in this study.

Symbol	Metric	Formula	Units
S	Slope	$= \mathrm{US}_{\mathrm{elevation}} / \mathrm{DS}_{\mathrm{elevation}}$	unitless
s	Sinuosity	$=L/L_s$	unitless
ε	Longitudinal roughness[a]	$=(\sum_{i=1}^{n} \lvert z_{\mathrm{obs},i} - z_{\mathrm{pred},i} \rvert)/n$	meters
X	Complexity	$=S \times s \times \varepsilon$	meters

[a]Where i is an interpolated point from the stream network segment between the US and DS stations, and $z_{\mathrm{obs},i}$ is the interpolated elevation along the stream network segment center line, and $z_{\mathrm{pred},i}$ is the predicted elevation of the observation at that point, given the mean slope of the reach.

Retention, R, described by Doyle (2005) is the amount of nutrient retained between an upstream concentration, C_0 (mg L^{-3}), and downstream concentration, C (mg L^{-3}). He proposed retention be defined as a portion of an incoming load, R, written as

R=1-(C/C_0). Basu *et al*. (2011) added the constraint that R must be a positive number greater than zero and less than one ($0<R<1$); values where $R<0$ indicates the channel is exporting, not retaining nutrients. Alternately, retention can be transformed to uptake velocity (c.f. Doyle *et al.*, 2003).

Uptake velocity, v_f, is a mass transfer coefficient that represents the rate nitrate moves down the water column (similar to a piston). It is empirically related to stream depth, h, and velocity, u. The three primary components of nutrient spiraling are the uptake velocity, v_f, areal uptake, U, and spiral length, s_w. These have a relationship that can be expressed in a mass balance equation (Newbold, 1992; Ensign and Doyle, 2006). In this context, v_f is independent of stream size but sensitive to concentration; as v_f approaches 0, ambient concentration and s_w increase. To calculate v_f in other studies, an isotopic tracer or nutrient addition has been used, which is costly and time consuming.

Rather than use a tracer in this study, I collected all of the parameters necessary for calculating uptake velocity based on equations from Stanley and Doyle (2002) and from Doyle (2005) that were

$$C/C_0 = \exp\left(\frac{-L\times v_f}{u\times h}\right)$$

I then modified, such that

$$v_f = -\frac{\ln(C/C_0)\times u\times h}{L}$$

Where:

h is the reach averaged depth,

u, the reach averaged velocity,

C, the DS concentration,

C_0, the US concentration, and

L, the reach length.

This equation addresses several challenges. The issue of reaches with different lengths is resolved given than uptake velocity is independent of length. One key assumption made in other studies (Stanley and Doyle, 2002; Doyle *et al.*, 2003; Doyle, 2005; Ensign and Doyle, 2006) is that transient storage has little influence over transport and retention; I am actually assessing its influence in this study. Another limitation of these models is that they are not spatially explicit models, which can be problematic as most biogeochemical processes are non-stationary at the scale of this study due to the heterogeneity of landscape. For instance, the amount of direct sunlight alters the net primary production and has a direct effect on vegetative uptake demand that can vary over short distances.

Statistical Procedures

Whenever v_f s were ≤ 0 they were changed to empty cells in SPSS for further analyses. Given that v_f s ≤ 0 were empty, SPSS ignored those cases in subsequent analyses. Using SPSS, I ran Levine's test of normality and a histogram to see if the parameters were normally distributed. Because the data were not in a normal distribution, I used the bivariate correlation function to run Spearman's rank correlation to calculate Spearman's Rho (α=0.05; two-tailed). This allows me to assess my hypotheses: first, nitrate rates will not correlate with channel complexity, and second, that complexity and land cover are equal determinants of spiral rates.

Results

Sampling and Laboratory Analysis

All reaches (n=31) were surveyed between 28 July and 9 November 2011. Sample times were distributed throughout the day (Table 2.2) with the earliest at 0800 h and the latest 1600 h. There could be potential bias due to different temperatures although it did not seem apparent from the results. The reach length varied from 6 to 50 meters with a mean length of 19 m, standard deviation of 11 m. Shorter reaches were typically located in lower order streams. The median sample date for the study was 30 October 2011. In total, 93 samples were taken from 31 reaches. To maximize hydrologic variability, sampling occurred during and after rain events (16 reaches, 48 stations). At 6 reach locations (18 stations) there was precipitation during fieldwork.

Table 2.2. Locations split by precipitation within 24 hours, precipitation is the average amount of precipitation recorded by HYDRA stations within 24 hours of sampling in centimeters. Reach temperature, date, time and nitrate concentration (µg/L) by station at each location listed

No rain	Date	Time	Temperature °C	Nitrate US[a]	Nitrate MS[b]	Nitrate DS[c]
0	7/28/2011	15:45	21.2	127	249	4
0	8/1/2011	14:00	18.8	418	14	410
0	8/8/2011	8:40	16.5	587	555	467
0	8/9/2011	10:00	14.8	364	517	470
0	8/10/2011	9:00	13.8	599	590	520
0	10/17/2011	10:50	10.3	1060	1040	1020
0	10/17/2011	13:00	11.1	726	776.5	782
0	10/20/2011	16:00	13.0	636	683	760
0	10/28/2011	8:45	10.4	177	121	303
0	10/28/2011	11:00	8.8	69.3	32.5	21.6
0	11/2/2011	9:45	5.8	662	651	852
0	11/2/2011	11:00	7.3	427	225	280
0	11/2/2011	13:00	9.0	1000	963	910
0	11/7/2011	12:15	9.1	551	585	538
Precipitation cm	Date	Time	Temperature °C	Nitrate US[a]	Nitrate MS[b]	Nitrate DS[c]
0.9	7/21/2011	13:30	13.1	494	484	476
9.1	10/11/2011	10:20	13.9	561	589	395
2.5	10/15/2011	12:30	12.0	1340	1130	1010
2.5	10/15/2011	13:00	12.3	1490	1640	1630
2.5	10/28/2011	15:00	9.5	110	87.5	110
4.4	10/30/2011	12:17	12.2	76.4	121	177
4.8	10/30/2011	14:20	13.0	222	200	280
1.7	10/31/2011	12:00	10.1	339	213	339
0.9	10/31/2011	14:00	9.5	580	608	580
0.1	11/3/2011	10:00	9.3	245	253	250
0.1	11/4/2011	11:10	11.7	1420	1440	1410
0.1	11/4/2011	12:40	7.4	2940	3080	2750
0.1	11/4/2011	13:45	9.9	689	696	228
0.4	11/6/2011	11:20	7.7	590	358	353
0.1	11/6/2011	14:00	10.0	420	173	449
0.8	11/9/2011	12:00	9.5	513	536	373
0.8	11/9/2011	14:15	9.1	532	476	443

[a]US = upstream
[b]MS = midstream
[c]DS = downstream

Discharge varied greatly (Table 2.3) with the highest value recorded during a storm event (*Q*=630 L s-1) at FU-08. Excepting storm events, discharge increased with stream order (Table 2.3). Interestingly, variation in discharge within reaches could be quite high (Table 2.3), which demonstrates the large amount of water entering and exiting the stream channel over very short distances. In addition, there was high variation in stream temperature ranging from 7°C to 25°C (Table 2.2).

Table 2.3. Reaches by station discharge, nitrate concentration, and location ID.

Q US L s^{-1}	*Q* MS L s^{-1}	*Q* DS L s^{-1}	NO_3^- US µg L^{-1}	NO_3^- MS µg L^{-1}	NO_3^- DS µg L^{-1}	Reach ID
0.89	0.17	0.51	494	484	476	SV02
42.25	19.68	27.19	127	249	4	AS01
1.70	1.50	1.18	418	14	410	VT04
28.20	17.81	14.22	587	555	467	FU11
11.35	8.82	0.93	364	517	470	FU14
5.24	18.70	0.89	599	590	520	VT03
355.10	559.31	630.90	561	589	395	FU08
1.62	1.54	2.40	1340	1130	1010	WD08
0.60	1.05	1.47	1490	1640	1630	AS11
1.92	2.16	1.80	726	776.5	782	VT06
4.52	2.10	4.20	1060	1040	1020	WD07
3.06	0.68	3.68	636	683	760	VT05
0.92	0.84	1.80	69.3	32.5	21.6	WD03
1.51	2.52	3.60	110	87.5	110	WD05
1.44	3.57	0.30	177	121	303	AS12
87.08	67.95	69.84	76.4	121	177	FU01
31.04	36.04	31.87	222	200	280	FU06
71.56	47.25	37.58	339	213	339	FU09
1.26	1.92	1.92	580	608	580	WD06
0.24	0.46	0.54	427	225	280	SV01
2.94	0.60	0.95	662	651	852	SV07
0.08	0.16	0.18	1000	963	910	VT02
3.68	2.59	3.53	245	253	250	FU04
3.45	4.62	3.86	689	696	228	FU15
1.02	1.92	1.04	1420	1440	1410	FU05
1.80	0.99	0.58	2940	3080	2750	FU18
44.70	51.84	85.50	420	173	449	AS03
57.48	53.52	56.95	590	358	353	FU16
5.04	5.38	5.54	551	585	538	AS13
11.04	1.23	1.38	513	536	373	AS10
1.20	1.29	1.58	532	476	443	VT01

Nitrate concentration generally decreased from the US location to the DS location (Table 2.3). Most reaches met the *R* ($0<R<1$) criteria (n=18). However, 3 reaches had a zero *R* value, which indicates they were in equilibrium; 10 reaches had positive *R* values, which means these reaches were net exporters of NO_3^-. Hence, in 18 locations I was able to capture the uptake velocity, and in 10 locations the spiral length was too long or short to capture uptake velocity. Given that the 10 reaches with positive *R* values, and the 4 reaches with a zero *R* value would result in negative uptake velocity estimates; these reaches were not considered in further analyses as has been done previously (Wollheim *et al.*, 2005; Newbold *et al.*, 2006; Claessens *et al.*, 2010). The average concentration of NO_3 was 614 µg/L and ranged from 4 to 2940 µg/L. The highest concentrations of nitrate were found in lower order streams, while the greatest change in nitrate levels occurred during a storm event (FU-16). For reference, Oregon has no total maximum daily load (TMDL) established for nitrate, as it is not considered a pollutant for non-drinking water. However the maximum contaminant level of nitrate in drinking water is 10 mg L^{-1}, which is a federal standard (United States Environmental Protection Agency, 2011).

Relative Contributing Areas and Complexity Index

Longitudinal roughness, ε, has the largest variability of the three complexity measures. Roughness values ranged between 9 – 1318 mm, with a variance of 0.12, the distribution of results was positively skewed. Many of the reaches surveyed were in Strahler first-order streams, which tend to have step-pool formations and steeper gradients than higher order streams (Table 2.4). In contrast, higher order streams tend to have greater sinuosity than lower order streams but less longitudinal roughness (Table 2.4). Generally, as ε values increase, so do X values, this is not the case for S and s values (Table 2.4); in other words as longitudinal roughness increases, so does complexity.

The complexity metric, X, ranged between 0.02 and 275.89 (10 x 1 m^{-3}). The distribution of complexity values had high kurtosis (17), mean and median X values were 23.83 and 3.38 (s.d. 53.08). Thus, complexity was highly variable among reaches of comparable length.

Table 2.4. Reaches by Strahler stream order, slope, sinuosity, *s,* longitudinal roughness, ε, complexity, *X*, reach ID, and reach length.

Order	Slope[a] x10^{-2}	s[a]	ε x10^{-2}, m	X, x10^{-3}, m	Reach ID	Length m
1	30.8	1.0004	20.8	64.1	SV01	6
2	3.02	1.0001	49.4	14.9	SV02	40
2	9.44	1.0053	132	125	WD08	30
2	0.21	1.0293	0.96	0.02	WD05	10
2	2.35	1.0672	2.51	0.63	FU04	20
2	0.56	1.2377	9.38	0.64	VT03	30
2	2.15	1.0172	54.1	11.8	AS11	10
2	2.82	1.0014	20.5	5.8	WD07	13
2	0.26	1.12	1.25	0.04	WD03	10
2	1.17	1	12.3	1.44	WD06	20
2	1.25	1.1908	5.9	0.88	VT01	9
3	0.56	1.004	5.48	0.31	AS12	10
3	21.6	1.0067	127	276	VT02	10
3	5.93	1.0231	71.9	43.6	VT04	30
3	6.55	1.0081	31.2	20.6	VT06	10
3	1.54	1.0443	10.8	1.74	VT05	15
3	4.26	1.0013	18.5	7.87	SV07	10
3	3.35	1.048	6.15	2.16	FU15	13
3	13.4	1.0309	15.5	21.4	FU05	10
3	12.4	1.0679	9.54	12.7	FU18	9
3	10.7	1.0242	18.1	19.9	AS13	14
3	13.1	1.0184	13	17.3	AS10	10
3	0.93	1.0417	16.2	1.57	AS01	50
3	7.23	1.0031	82.9	60.1	FU14	47
4	0.23	1.0307	0.94	0.02	AS03	15
4	0.2	1.3657	15.4	0.43	FU11	20
4	7.16	1.1226	2.53	2.65	FU08	20
4	1.58	1.0474	12.8	2.13	FU06	20
4	1.47	1.1379	27.6	4.62	FU09	30
4	1.08	1.0245	4.75	0.53	FU16	15
5	0.66	1.0059	9.43	0.62	FU01	20

[a]Unitless

Statistical Procedures

Given that the data were not normally distributed, the non-parametric Spearman's rank correlation matrix was used to assess the relationship among variables (Table 2.5). The strongest negative correlation was between uptake velocity, v_f and longitudinal roughness, ε (-0.676, α=0.002) indicating that the rate of nitrate uptake (the uptake velocity, v_f) decreases as longitudinal roughness increases. Also, a significant negative correlation was discovered between uptake velocity and both slope and longitudinal roughness (Table 2.4). Slope, longitudinal roughness, and complexity all show positive correlations with nitrate concentrations at each station, which means nitrate concentrations are higher where there is greater slope and longitudinal roughness. Average nitrate concentrations are negatively correlated with uptake velocity (-0.416, α=0.086). While middle and upper reach stations are also negatively correlated, they are not statistically significant (Table 2.4), although this may be a function of the method used to calculate uptake velocity. Discharge is significantly, strongly, positively correlated with uptake velocity, at all stations, and with the reach averaged-discharge values, hence high roughness decreases discharge which lowers uptake. Precipitation events within 24 h show a weak positive correlation with uptake velocity, (0.215, α=0.393, n=18). However, one caveat of this study is that each location was only visited once, hence meteorological conditions were not controlled for and could be a factor that affects uptake velocity. Nevertheless, be sampling upstream and downstream of each reach I am able to quantify the spiraling taking place during that visit regardless of conditions external to the reach itself.

In general, there were several trends observed. First, increases in discharge resulted in lower concentrations but greater variability of NO_3^-. Second, when complexity increased the concentration of NO_3^- also increased, but uptake velocity was lower. Third, as NO_3^- concentration decreased, so did uptake velocity, and these both correlated with an increase in wetland and grassland area.

Table 2.5. Spearman's rank correlation matrix, values significant at the 0.05 level (2-tailed) or better shown below in bold

	V_f	Average NO_3^-	Average Q	S^a	s^b	ε^c	X^d	Grass	Wetland
V_f Sig. *n=18*									
Avg. NO_3^- Sig. *n*=31	-.416 .086								
Avg. Q Sig. *n*=31	**.552** **.018**	**-.361** **.046**							
Slope Sig. *n*=31	-.406 .095	**.494** **.005**	**-.492** **.005**						
Sinuosity Sig. *n*=31	.414 .088	-.156 .403	.308 .092	**-.390** **.030**					
ε Sig. *n*=31	**-.676** **.002**	**.434** **.015**	**-.355** **.050**	**.651** **.000**	**-.475** **.007**				
X Sig. *n*=31	**-.610** **.007**	**.510** **.003**	**-.480** **.006**	**.927** **.000**	**-.431** **.016**	**.870** **.000**			
Grass Sig. *n*=31	.404 .097	**-.458** **.010**	**.380** **.035**	**-.539** **.002**	.058 .757	**-.354** **.050**	**-.525** **.002**		
Wetland Sig. *n*=31	.404 .097	**-.444** **.012**	**.375** **.037**	**-.540** **.002**	.053 .775	**-.356** **.049**	**-.526** **.002**	**.999** **.000**	

[a]slope
[b]Sinuosity
[c]longitudinal Roughness
[d]complexity metric

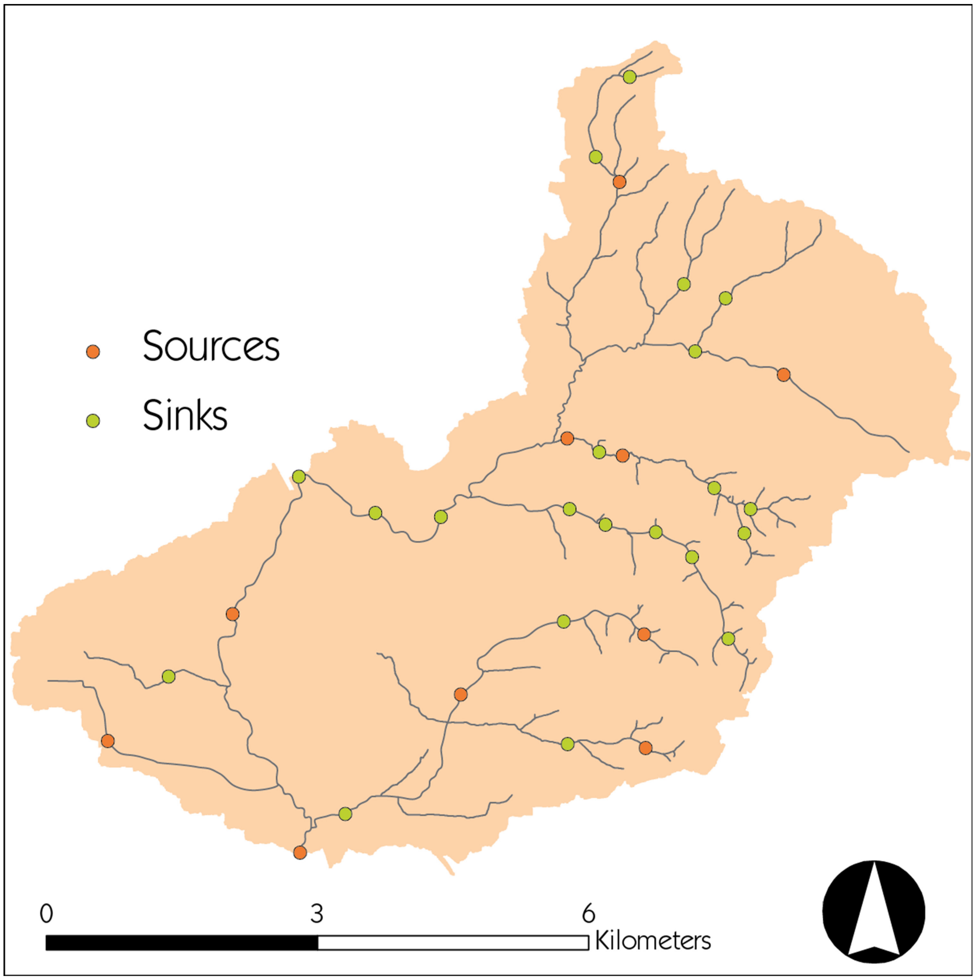

Figure 2.5. Reach locations in green where retention was detected (sinks) and valid spiral metrics computed. Areas that were sources are in orange and no spiral metrics were computed.

Discussion

Few, if any, links appear to exist between land use and complexity, nitrate uptake, or nitrate concentration. Channel complexity indices show a high level of heterogeneity in the studied reaches despite the fact that the land use has been classified as relatively homogenous. The sample results demonstrate that there is nitrate uptake occurring in upper Fanno Creek watershed, and given that channel complexity is correlated with nitrate uptake, it is evident that the channel itself is processing this nitrogen.

Land Use Correlation

In existing studies, Fanno Creek watershed has been characterized by homogenous, predominantly urban land use (Chang, 2007; Duh *et al.*, 2008; Chang *et al.*, 2010; Jung *et al.*, 2011). In this study, some variation in land cover was calculated in reach contributing areas—up to 40% more forested land in some areas. The spatial distribution of forested land is consistent with the watershed topography, in that forested land is highest where slopes are steepest. My observations in the field noted differences in how the same land use type (and even same the zoning code, e.g. single-family residential) varied considerably in management. For example, site WD-03 (Figure 2.6, 2.8), on the right bank, is a single family residence with a lawn extending to the wetted perimeter while at site SV-07, residents of single family homes around the stream have attempted to create a park-like environment (Figure 2.7, 2.8). While not suggesting grass lawns are the sole cause of elevated nitrate levels, fertilized lawns are capable of contributing nitrate. When lawns are mowed, the grass clippings can introduce additional nitrate to the stream. At other locations such as SV-07 with natural landscaping, nitrate concentration may be lower. Surprisingly, SV-07 revealed higher NO_3^- concentrations than WD-03 (Table 2.3) despite greater complexity (Table 2.4) and a visibly larger riparian buffer.

Despite a low complexity value at WD-03, it may be that the horizontal complexity—sinuosity—was actually high but too small to be captured by LiDAR. The grassland had a large floodplain that it appeared to be actively connected to, evidenced by the bank failure and erosion seen when surveying; though it might have much more transient storage than the more forested site, SV-07. Discharge in WD-03 was far less variable than SV-07 and combined with low nitrate values could be evidence of more subsurface flow. The variable discharge in SV-07 can be attributed to transient storage with short residence times, which result in little reduction of nitrate.

Figure 2.6. Photo of WD-03, Woods Creek (Hideaway Park on Left); arrow indicates the direction of flow.

Figure 2.7. Photo of SV-07, Sylvan Creek, semi-private park; arrow indicates direction of flow.

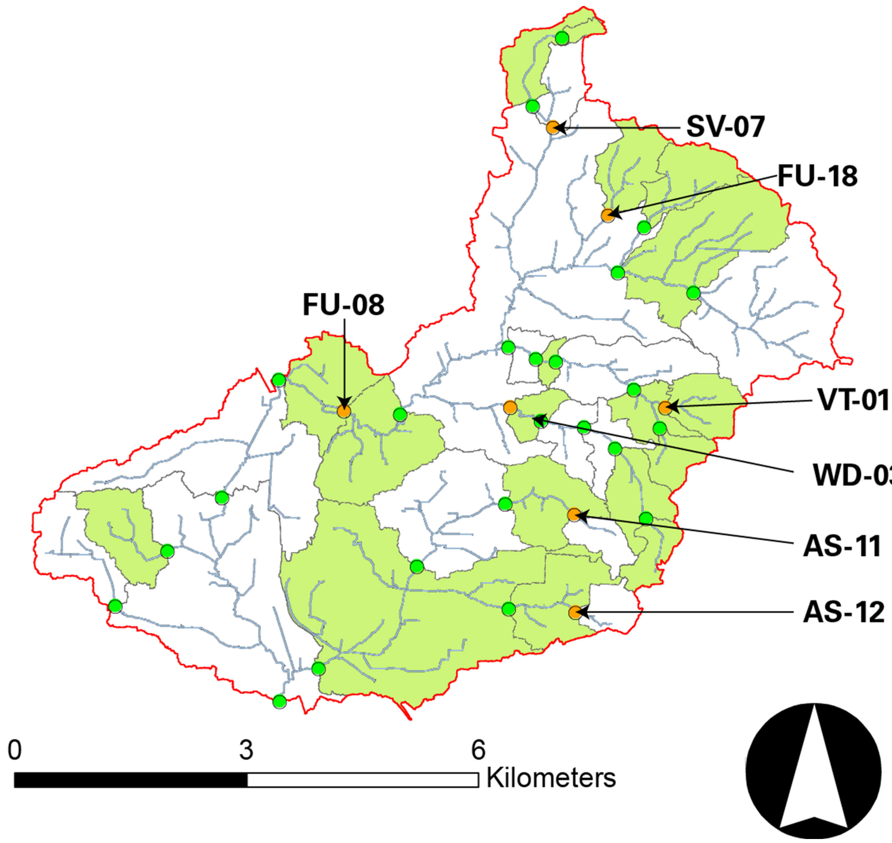

Figure 2.8. Location of reaches mentioned in this paper, green areas represent upstream contributing areas with retention

Complexity Values

Complexity metrics were highest in the headwater, higher elevation locations of the watershed. Thus, the strongest negative correlation in this study was between discharge and uptake velocity, given that complexity values reflect geomorphic units. A step-pool morphology common in headwater reaches will have a stronger effect on discharge than a meander belt in a larger reach. This correlation (-0.501 Spearman's ρ, α=0.040) demonstrates that vertical complexity (e.g. step-pools) is equally, if not more important than horizontal complexity (e.g. meander belts). Increased flow moving through transient storage is a characteristic of first-order streams with high vertical variability and relatively short residence time. Higher order streams with more horizontal (sinuous) complexity have more transient storage and longer residence time which can potentially lead to more biogeochemical processes, such as nitrate removal (Gooseff *et al.*, 2008).

In an urban watershed, such as Fanno Creek, characteristic "flashy" discharges would be expected in lower order streams more frequently than "natural" streams of the same order. If discharge has a dramatic effect on uptake velocity, then it stands to reason that an urban stream will not have the same response as a natural stream without some type of management, restoration, or mitigation that reduces the flow.

Location FU-08, a 3rd order reach on the main stem of Fanno Creek, showed the highest recorded discharge values in this study (Table 2.3; Figure 2.8, 2.9, 2.10, 2.11). Given that discharge was high, and that in other studies increased discharge generally lowers uptake velocity, it was unexpected that this reach uptake velocity value was also moderately high (v_f=384.66 mm min^{-1}). Channel complexity may play a role in explaining the considerable amount of NO_3^- lost between the US and DS sections at this location. A comparison of cross-sections (Figure 2.10) confirms a large increase in channel depth at the middle section. This increased average depth raises the uptake velocity due to slowing of the surface water velocity. Here, longitudinal roughness may also be hydrogeomorphic driver (Figure 2.11) as there is evidence of scour before the downstream station. A bridge and bank armoring downstream may have artificially created transient storage in the form of dead storage at this location (Gooseff *et al.*, 2008). This is indicative of a hot spot, or hot moment, for nitrate removal during storm events (Vidon *et al.*, 2010). Further testing would be necessary to determine if the same location would be a hotspot for nitrate removal under baseflow conditions. Slowing flow may not be desirable in some areas prone to flooding, but may nevertheless be a design consideration when restoring streams. This study illustrates how structure is capable of influencing both hydrology and biogeochemical function simultaneously.

Figure 2.9. Photo of FU-08, Fanno Creek, highest discharge with high uptake velocity, the arrow indicates direction of flow.

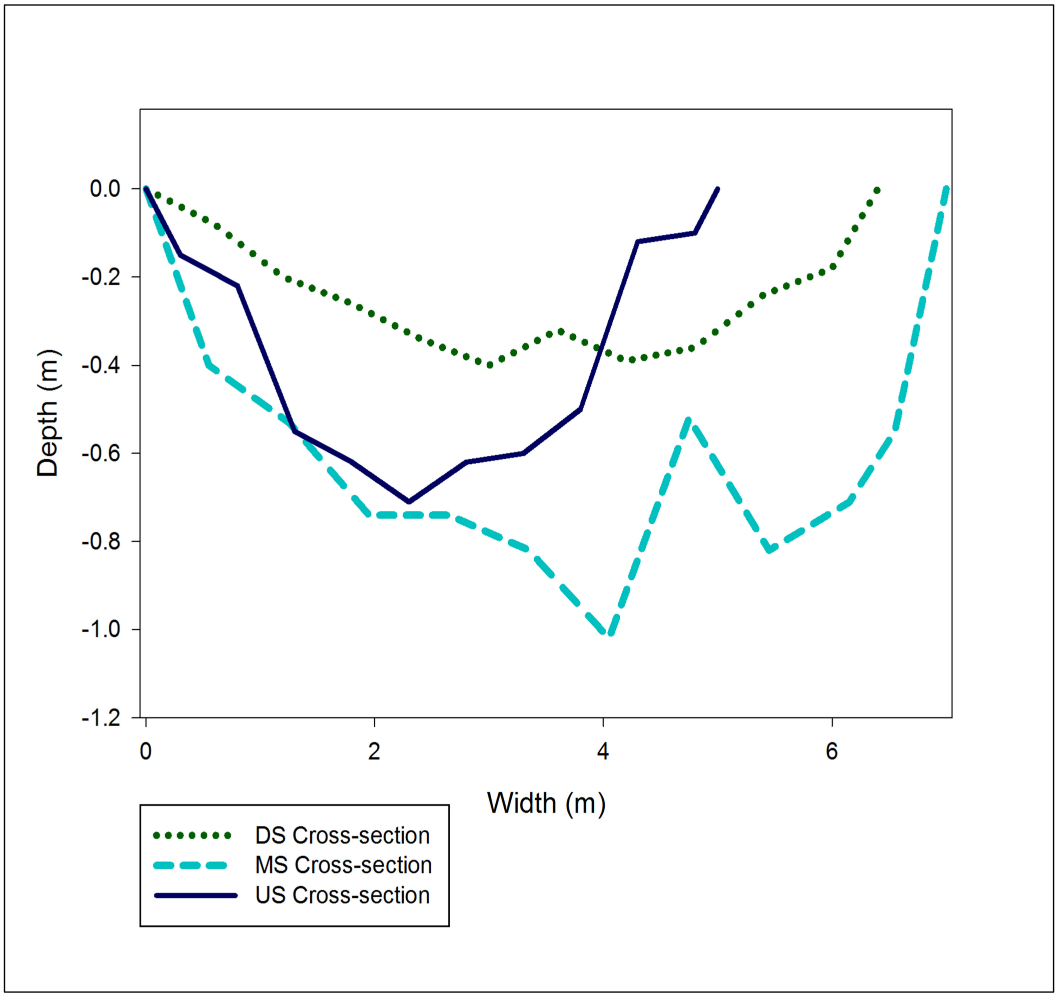

Figure 2.10. Comparison of downstream (DS), midstream (MS), and upstream (US) cross sections areas at location FU-08.

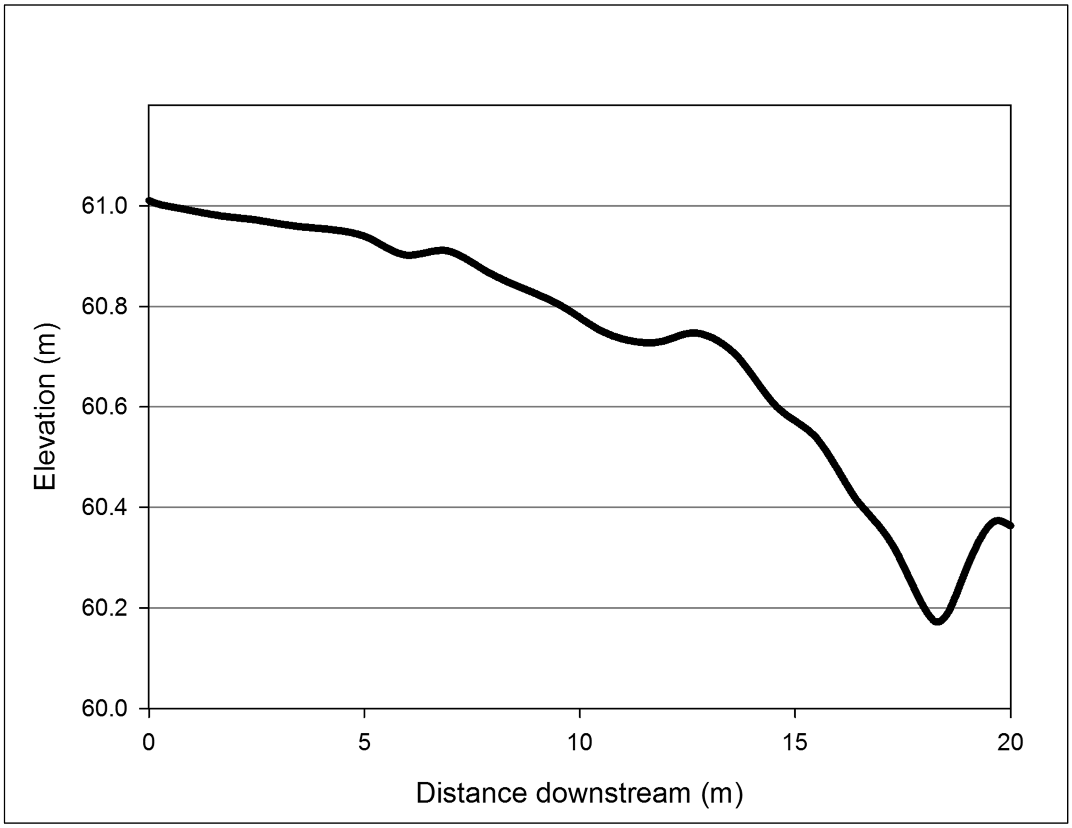

Figure 2.11. Longitudinal profile of FU-08 channel center line.

Relationship of Nitrate and Complexity

One reach with low concentration of nitrate and no retention was in a headwater reach, AS-12 (Figure 2.8, 2.12). The City of Portland (City of Portland Bureau of Environmental Services, 2011) designated AS-12 as a water quality treatment area. The reach is buffered considerably by a park with no trails or nearby paths. This same reach had no retention of nitrate; in fact, nitrate concentration nearly doubled between the upstream and downstream station in the reach (177 $\mu g\ L^{-1}$ US–303 $\mu g\ L^{-1}$ DS). The complexity value for AS-12 (3.10E-04 m) was nearly an order of magnitude lower than FU-08 (2.65E-03 m), despite the fact that AS-12 was a restoration site, whereas FU-08 was not. The nearby headwater stream reach AS-11 (Figure 2.8) showed much higher complexity (2.15E-02 m) but no retention. NO_3^- at AS-11 was much higher than AS-12, ranging from 1490 μg L^{-1} US–1630 μg L^{-1} DS. Both AS-11 and AS-12 had similar *Q* values; at AS-12, Q ranged from 0.6-1.5 L s^{-1}, and at AS-11 it ranged from 0.3-1.4 L s^{-1}. Thus, headwater reaches of the same watershed can have high and low complexity, high and low nitrate concentration, and no retention at either site. This demonstrates how sites that may appear equivalent are not.

Sites may have similar type locality like AS-11 and AS-12 with the same potential natural ecosystem service, yet they are not functional due to other effects related to upstream or lateral contribution of nitrate. Thus, impaired function can be the result of land use, so it is important to realize that complexity alone is not the only driver of biogeochemical function. Discharge also plays a role, but is not the sole determining factor in whether a reach will have uptake function.

Not all headwater streams were like AS-11 and AS-12; in contrast, VT-01 (Table 2.3; Figure 2.8) had a complexity value lower than AS-12 (8.76E-04 m), a similar range of *Q* (from 1.20 L s^{-1}–1.58 L s^{-1}), and nitrate concentration ranging from 532 μg L^{-1} US–443 μg L^{-1} DS. It is surprising then that VT-01 had retention despite its similarities with AS-11and AS-12. In riparian areas, nitrate is tightly cycled, indicating such a strong vegetative demand that nitrate may not travel very far downstream before it is taken up for consumption (Schlesinger, 1997). The concentration of nitrate influences uptake velocity, and, with higher demand for nitrate in a riparian forest, these areas should have tighter cycles (and higher levels of nitrate) than those with less vegetation. In another study, Claessens *et al*. (2010) determined that transient storage and saturation conditions exerted a negative effect on v_f, and that urban streams in Baltimore were exporting nitrate as a result.

Figure 2.12. Photo of AS-12, Ash Creek, natural water quality area in Dickinson Park, Portland, Oregon; arrow indicates direction of flow.

Uptake velocity and stream order are significantly, positively correlated (0.803 Spearman's ρ, α=0.00, n=18). As stream order increases, their uptake velocity also increases. This is expected as other studies (Ensign and Doyle, 2006) have shown that as stream order increases, uptake velocity also increases, generally peaking at the 3rd or 4th order and then declining. Tank *et al*. (2008) comprehensively reviewed spiral studies done since 1990, which combined with the review by Ensign and Doyle (2006), presents a large database of uptake velocity measurements and NO_3^- concentration. In Figure 2.13, I compare these studies to urban spiral studies in Arizona (Grimm *et al.*, 2005), Baltimore (Claessens *et al.*, 2009), and this research. A clear distinction is evident between uptake velocity and nitrate concentration in Arizona, Baltimore and Portland. Portland has relatively higher nitrate uptake velocity than Baltimore and Arizona, but lower concentrations (Figure 2.13). This means that although there have not been many urban spiral studies, there seems to be a difference between Arizona, Baltimore and Portland in terms of the concentration and rate of uptake velocity. This could mean the functions used to model spiraling in natural systems may be different in urban areas. More work should be done to examine seasonal, geographic, and geomorphic variability in spiraling. Spatially explicit methods such as the one used in this study might help us better understand landscape variations in biogeochemical function, which could lead to better management practices and healthier streams.

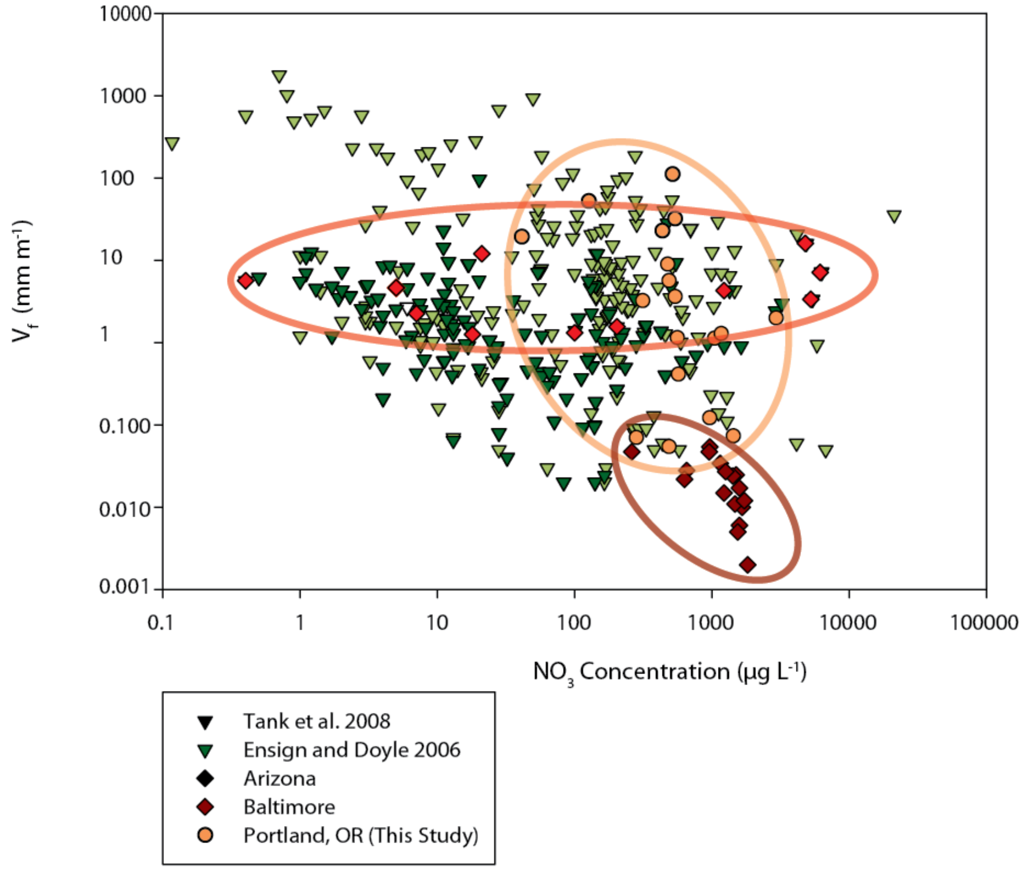

Figure 2.13. Comparison of nitrate uptake velocity in other studies (Ensign and Doyle, 2006; Tank *et al*., 2008) and this study; the uptake velocity values from this study are much higher than studies in Arizona (Grimm *et al*., 2005) and Baltimore (Claessens *et al*., 2009). This study had 31 reaches, while the Arizona study used 10 and the Baltimore study used one 2.4 km reach.

Wetlands and grasslands are negatively correlated with complexity, which may be a geomorphic response to land use change. Areas like WD-03, discussed earlier (Figure 2.6, 2.8), had low complexity and a grass (lawn) that extended to the wetted perimeter. Clearly sediment loads had increased due to bank failure, which may have helped to straighten the channel (Knighton, 1998). Incision caused by high magnitude discharges frequently observed in Fanno Creek help to explain bank erosion, which ultimately contributes more sediment that aggrades in some reaches while scouring others. Several studies observed this kind heterogeneous response in an urban context (Chin, 2006; Gregory, 2006).

The scale of this study may have been too small to capture higher-order stream complexity (e.g. sinuosity on a 3rd order stream might have a meander belt several hundred meters long). Other factors such as time of day and precipitation did not have any significant correlations with this study while other studies (Bosch, 2008) have shown that nitrate concentration is diurnal and that stormwater also has different chemistry than surface water at base flow (Chang and Carlson, 2005). I did not compare stream water to stormwater chemistry; however, my results did not indicate any significant correlation among nitrate concentrations, uptake velocity, the time, or amount of preciptiation. Hence in this study, precipitation did not have a strong or significant correlation with whether or not a site would have retention of nitrate.

This study was composed of reaches in predominantly 1st and 2nd order streams. Ensign and Doyle (2006) noted that as stream order increases to the 4th order, so does nitrate uptake velocity. The finding here are consistent with their study, given the positive correlation between discharge and uptake velocity. One explanation for this correlation is that as stream order increases, so does discharge. Further, as stream order increases, slope and longitudinal roughness will decrease, and sinuosity should increase. Due to the fact that most streams were 1st and 2nd order, complexity values have more correlation with longitudinal roughness and slope than with sinuosity, which explains why longitudinal roughness and slope have stronger correlation with complexity than sinuosity. It would be useful then to look at an equal number of 1st, 2nd, 3rd and 4th order streams to see if longitudinal roughness and slope are as strongly correlated with uptake velocity.

Conclusions

My hypothesis, that complexity and uptake velocity are correlated, was confirmed as 18 out of 32 sampled reaches measured showed uptake velocity. Land use did not show a significant correlation with complexity. Lack of complexity is a characteristic often attributed to urban streams and a driver of the urban stream syndrome, due to the fact that less complex streams have flashy discharge. It is not surprising then, that discharge had strong negative correlation with complexity. This study emphasizes the necessity of further research in urban watersheds at multiple scales to establish relationships between structure and function with respect to biogeochemistry cycling. Further, it demonstrates the utility of a method, using ion-selective electrodes and LiDAR, to quickly assess and monitor an area at a high spatial-temporal resolution. Finally, in the context of river restoration and research, it illustrates how the structure of a reach is correlated with more than just hydrogeomorphic variables.

Future work might re-examine locations from this study, given that this research updates an existing study from 2000. Alternatively, a few sites could be visited repeatedly over the course of a year in order to conduct a synoptic sampling that might identify a site as a hot spot or hot moment for a given discharge. This study provides a seasonal range of nitrate values that would be necessary for further nitrate spiral tests, including those using additions or isotopic tracers. Recognizing that urban streams have many different drivers, more studies are needed to show how in what ways they function differently than "natural" riparian ecosystems (Grimm *et al.*, 2005; Claessens and Tague, 2009; Baker *et al.*, 2012). This implies that although it might not be possible to restore an urban riparian system to a natural functioning state; we can manage urban riparian systems based on desired outcomes (e.g. water quality, habitat, flood control). Hence, channel morphology can synergistically create responses that allow multiple desirable outcomes to be achieved simultaneously.

There is a continual need evaluate the connection between geomorphology and uptake velocity (e.g. Claessens *et al.*, 2009; Stewart *et al.*, 2011; Baker *et al.*, 2012). Although challenging to measure, transient storage is a crucial driver of hot spots and hot moments, and accordingly should be an essential tool in integrated watershed management. Complexity alone does not quantify the innate functional capacity of a stream for N-processing. Complexity combined with other measurements like water quality (e.g. nitrate, temperature) and flow, which is subsequently integrated into a network, creates the manageable goal of using streams themselves as tools for creating future restoration, rehabilitation and research opportunities. Big changes can occur at a small scale. Thus, two or more water quality measurements should be taken when visiting monitoring sites.

The research presented embodies a first-step in a series of investigations which strive for a comprehensive understanding of the structural-functional relationship between hydrogeomorphology and biogeochemistry. Moreover, it proposes consideration for integrating multiple, advantageous strategies into future restoration work. The importance of scalar issues in consideration of sampling protocol cannot be overstated. Complexity alone may not be the solution to water quality problems, but it offers insight into a means for enriching outcomes.

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