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Evaluation of the Relationship Between Land Use and Water Quality in Kittitas County, WA

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EVALUATION OF THE RELATIONSHIP BETWEEN LAND USE

AND WATER QUALITY IN KITTITAS COUNTY, WA

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by

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ABSTRACT

EVALUATION OF THE RELATIONSHIP BETWEEN LAND USE

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Water in Kittitas County is extremely valuable since it supports farming, recreation, and cultural activities, as well as environmental processes and a diversity of biological life while providing many ecosystem services. However, land conversions required by agricultural and urban land uses can negatively impact water quality and the biological function of the stream. I studied how forested, agricultural, and urban land use affect six streams. Fourteen sites were sampled, once each in July, August, and September 2019. Land use was calculated as a percentage of forested, agricultural, and urban land use within a 100-m buffer of the stream, upstream of the sample site. Measurements of the streams at the sample sites, including thalweg depth, discharge, bank full width, and a substrate analysis, were taken as well as temperature, pH, and dissolved oxygen. Suspended sediment, specific conductivity, and turbidity were also determined, and samples were collected to measure ammonium, nitrate, and phosphate concentrations. An analysis of EPT percentage and HBI scores for aquatic benthic macroinvertebrates were used to infer biological condition. I found that land use had a significant effect on depth, discharge, temperature, specific conductivity, nitrate, phosphate, EPT, and HBI. Agricultural and urban land uses had deeper channels with high flows, and high temperatures. Temperatures in agricultural and urban land uses never went below 13°C and had the highest peak at 21°C, while forested sites had a low temperature at 10°C and never went above 14°C. Also, I found that nitrate and phosphate concentrations, as well as HBI, were highly correlated with a higher percentage of agricultural and urban land use. High EPT percentages were highly correlated with high forested land use. Management recommendations include best management practices (BMPs) for different agricultural and urban sites. These BMPs are targeted to reduce nutrient inputs and increase habitat heterogeneity for the restoration of sensitive macroinvertebrates. Overall, this study highlights how land use is associated with degraded stream habitat showing the biological consequences observed in the aquatic macroinvertebrate community in Kittitas county.

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I would also like to thank WA Department of Ecology (DOE) which sparked my interest in water resources in arid environments. Specifically, I would like to thank Candis Graff, Permit Writer with DOE for believing in me and for being a great mentor.

This study would not have been completed without the help of my husband David Schulz. Thank you for all your help from sampling with me when I did not have a research assistant to providing a stable and loving home. I would also like to thank my father and mother, Robert and Tabitha Wood, for their love and support.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF FIGURES (CONTINUED)

LIST OF FIGURES (CONTINUED)

CHAPTER I

INTRODUCTION

Research Problem

Converting open land to agricultural or urban land uses and maintaining those land use practices can negatively impact stream water quality and biological function. Even though land use conversion is required for crops and residential/commercial needs, stream systems can provide valuable ecosystem services that make protection of water quality an important societal goal (Foley et al. 2005). To that end, the United States Congress passed the Clean Water Act in 1972, requiring streams that do not meet minimum standards of water quality to be listed under section 303(d). Once a stream is listed, states must develop a Total Maximum Daily Load report requiring the use of "all existing and readily available information" on stream water quality (40 C.F.R. §130.7(B)(5)) for the purpose of improving water quality. In a mixed land use watershed, getting an accurate picture of stream water quality can be costly and time consuming because land use differences over relatively short distances can influence water quality with consequences for stream biological function. Moreover, land use effects on water quality vary among watersheds globally and regionally, so predicting impacts is difficult (Regetz 2003; Foley et al. 2005; Conway 2007; Tu et al. 2007; Jorgensen et al. 2009; Fiquepron et al. 2013; Tu 2013).

With the goal of protecting and improving water quality, citizens and political organizations collaborate to improve stream health, water storage, and stream habitat in Kittitas County. For example, in 2018 $1.4 million was allocated through grants to the Kittitas Conservation Trust for water quality improvement projects and floodplain management (Holappa 2018). This money was used for stream restoration projects located on Box Canyon Creek, the Upper Yakima River near Cle Elum, Gold Creek, and the Upper Kachess River (Holappa 2018). Past projects in the Kittitas Valley include consolidating an irrigation diversion on Manastash Creek to increase instream flow (WA State Recreation and Conservation Office 2014) and restoration of the Reecer Creek floodplain to improve instream habitat (Mid-Columbia Fisheries Enhancement Group). The projects in Kittitas Valley are important because the arid shrub-steppe lands contain critically important agricultural lands that require irrigation water, but the streams also support endangered species of salmon. Although water quality is an important management goal for diverse interest groups (Dittmer 2013; Macfarlane et al. 2017; Office of Columbia River 2018), an analysis of water quality and how different land uses may influence it has not been completed in the Kittitas Valley.

Purpose

The purpose of this study was to correlate stream water and habitat quality data with dominant upstream land use at multiple points in streams draining through Kittitas Valley. Kittitas Valley stream systems are ideal because of easy access to many streams with a clear land use gradient and minimal inter-site variability. Water quality indicators were measured in accordance with Washington State Department of Ecology (DOE) protocols (WA State Department of Ecology 2019). The objective of this study was to use a whole system approach to create a comprehensive evaluation of stream health at each sample point to evaluate which stream sections are more degraded than others. This whole system approach included various methods to evaluate biotic and abiotic factors. Additionally, the data from this analysis has been made available to Department of Ecology, as well as a University of Arizona based group called Collaborative for Research in Arid land Stream Systems, whose goal is to compile stream data in arid land stream systems. Finally, this analysis includes management recommendations on stream sections that are identified as in need of restoration to support the development of a management plans by policy makers and public stakeholders.

Significance

Stream health and high standards of water quality are valued for many important reasons (Office of Columbia River 2018; The Yakama Nation 2019). All aspects of the stream system are culturally important to the Yakima Nation, who have many sacred uses for them that require the maintenance of high water quality (The Yakama Nation 2016, 2019). High water quality is also important in supporting the recreation value of this area as many people come to this area to fish, swim, or float the Yakima River. Cattle herds rely on the water in these streams as do the farmers irrigating crops. There is also value in a healthy ecosystem's ability to support a diversity of life as well as provide ecosystem services (Foley et al. 2014). Although these cultural values might seem disparate, they all share a common need for clean water, which makes this study important.

CHAPTER II

LITERATURE REVIEW

Forested, agricultural, and urban land use activities can affect water quality of stream systems in a variety of ways (Peters E. and Meybeck 2000; Russell et al. 2001; Regetz 2003; Williams et al. 2005; Conway 2007; Tu et al. 2007; Tu and Xia 2008; Jorgensen et al. 2009; Li et al. 2009; Tran et al. 2010; Fiquepron et al. 2013; McDowell et al. 2017). Compared to other land uses, forested streams generally have better water quality because they are typically in recreational and/or conservation areas that have little development and less intense land uses (Fiquepron et al. 2013; Tu 2013). These streams tend to have more riparian tree cover shading the stream, keeping temperatures cool, and stabilizing the banks. Streams in forested areas are typically not channelized and are usually not affected by irrigation withdrawal or return flow, allowing for more heterogeneity of habitats within the stream (Negishi et al. 2002; Allan 2004; Schroder et al. 2016). However some forests support logging, which can degrade water quality due to increased suspended sediment levels from erosion or increased stream temperature caused by reduced riparian canopy cover, both of which can negatively affect salmonid health (Gibbons DR 1973; Gregory et al. 1987; Chamberlin et al. 1991). Selective or minimized logging can reduce these negative effects (Cassiano et al. 2020).

In contrast to forested streams, agricultural streams frequently have poor water quality (Russell et al. 2001; Woli et al. 2004; Tu and Xia 2008; McDowell et al. 2017).

Intensive livestock farming introduces significant amounts of nitrate into stream systems (Woli et al. 2004), and livestock farming near small streams that lack fencing causes downstream accumulation of pollutants in ecosystems (McDowell et al. 2017) even when the proportion of agricultural land use is about the same in small and large streams (Williams et al. 2005; McDowell et al. 2017), illustrating the importance of near stream activities on water quality. Beyond livestock impacts, cultivated lands also can degrade streams. Lack of riparian cover common in agricultural streams can increase stream temperature (Younus et al. 2000), and agricultural land can increase specific conductivity in streams (Dow and Zampella 2000), both of which cause a decline in benthic macroinvertebrate populations (Jorgensen et al. 2009; Suter and Cormier 2013). Nitrate concentration frequently increases as agriculture land use increases (Wernick et al. 2007), sometimes exceeding the national drinking water standard and requiring the need for purification (Hatfield et al. 2009). Also, non-point sources of suspended sediment contribute to 34-65% of the sediment load in agriculturally-dominated watersheds (Russell et al. 2001). Suspended sediments have been widely studied as a cause of poor water quality because they carry fertilizer and pesticide pollutants into stream systems (Waters 1997; Cassiano et al. 2020). Therefore, suspended sediment loads can also indicate non-point source pollution loads (Gao 2008; Chang et al. 2013). Suspended sediments reduce light penetration to the stream bottom, affecting primary production and food web productivity by smothering vegetation (Clark II et al. 1985). Excess suspended sediments also damage fish and invertebrate gills, and the settling of fine particles can impact substrate conditions, decreasing habitat availability for aquatic species that require interstitial space between particles (Lauver 2012; Relyea et al. 2012). Invertebrate communities that need heterogeneity in the stream substrate for protection and laying eggs are also affected by the channelization of agricultural streams (Potyondy and Hardy 1994; Negishi et al. 2002; Kusnierz and Holbrook 2017), which leads to increased water velocity, erosion, decreased substrate size, and ultimately downcutting (Pedersen et al. 2014). These changes in substrate size could explain why trout populations are smaller and individual fish have smaller average length in channelized streams (Duvel et al. 1976).

Streams in urban areas display a pattern of ecological degradation known as "urban stream syndrome" (Walsh et al. 2005). Urban areas have a larger amount of impervious surface cover than other land uses, which causes increased non-point source pollution during precipitation events (Walsh et al. 2005). Urbanization and impervious surfaces also degrade water quality and stream health (Walsh et al. 2005; Conway 2007; Tu et al. 2007; Jorgensen et al. 2009) by increasing stream temperatures, water and pollutant runoff, and fine sediment delivery to streams (Conway 2007). Stream temperatures also increase due to lack of riparian cover as well as the "heat island" effect often found in urban areas (Paul and Meyer 2001; Walsh et al. 2005). Non-point source pollution from calcium carbonate weathering of concrete in urban areas can also increase pH and conductivity (Conway 2007; Tu et al. 2007; Jorgensen et al. 2009). Increased impervious surfaces and lack of adequate riparian cover can also increase suspended sediment (Walsh et al. 2005). Some studies also suggest that small streams with about the same proportion of urban land as larger streams have disproportionately more water quality degradation (Williams et al. 2005; McDowell et al. 2017), suggesting that small streams with small urban footprints may be more susceptible to urban impacts.

Because urban and agricultural land uses have both been shown to decrease the richness of benthic macroinvertebrates (Paul and Meyer 2001; Allan 2004), they are commonly used to study water quality because they have high and predictable sensitivity to water quality degradation (Li et al. 2010). The order plecoptera has been shown to be the most sensitive to organic pollution and stream degradation (Figure 1A). The orders trichoptera and ephemeroptera have also been shown to be sensitive to organic pollution (Figure 1). There is a well-documented decrease in these orders as pollution increases because at some point in the life cycle these macroinvertebrates were not able to survive the stream conditions (Hilsenhoff 1988). Moreover, because they do not readily move along the reaches of the stream, using them to indicate water quality allows for site-specific determinations required for many studies (Watershed Science Institute; Lenat 1988; Early et al. 2002; Kitchin 2005; Relyea et al. 2012). Agricultural and urban land use has been shown to drive ecological simplification within stream systems (Peipoch et al. 2015). This can lead to a reduction of landscape complexity and ecological integrity. Structural changes within the stream system from human land use, including channelization, have been found to increase ecological simplification. The consequences of this are a loss of heterogeneity and loss of biological function (Peipoch et al. 2015).

(A) (B) (C)

Figure 1 Common macroinvertebrates used to study water quality. (A) Plecoptera perlodidae. (B) Trichoptera limnephilidae. (C) Ephemeroptera euthyplociidae.

There is a link between the complexity and integrity of floodplains, so to understand how to best restore a stream, it must first be assessed to see what condition the stream is in (Figure 2). Typically, agricultural and urban land uses disconnect a stream from the natural floodplain, leading to the need for restoration if conservation is the goal.

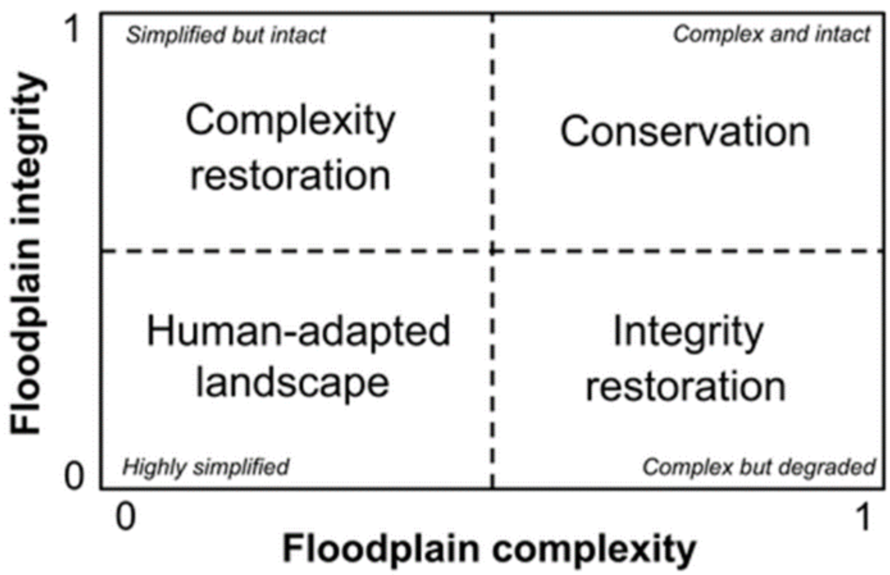

Figure 2 Riverine landscape condition. Describes how stream systems go from simplified to restored. Source: Peipoch et al. (2015).

Insect community condition can also be used as a response variable to measure the efficacy of stream restoration projects, which have increasingly been used to mitigate land use degradation of water quality (Bernhardt et al. 2005). However, neither sensitive species nor water quality tend to respond positively to restoration projects (Moerke and Lamberti 2004), likely because the chemical, hydrological, and physical elements of streams are not being altered enough to restore water quality to a level that would support sensitive species (Bernhardt and Palmer 2018). Many positive effects of stream restoration on biotic communities are short term and confined to the restoration site (Feld et al. 2011) with some long term positive effects on macrophytes that are still confined to the restoration site (Lorenz et al. 2012). This problem could be due to poorly executed restoration that lacks knowledge about chemical, physical, and hydrological alterations needed for success. For example, if substrate in restored reaches of a stream has unnatural placement, macroinvertebrate diversity can actually decrease (Pedersen et al. 2014).

Stream health is extremely important for many different reasons including subsistence and recreational salmon fisheries. Up to 70% of the water quality of high order streams is determined by head water or low order streams of that water shed (McDowell et al. 2017). These higher order streams, including the Yakima river, support migratory fish and must meet certain water quality conditions for their success, including proper substrate types for laying eggs, providing food for juvenile fish, and regulating temperature in summer heat (Jorgensen et al. 2009). Water quality degradation can increase fish mortality rates from parasites and disease despite having good quality habitat in other respects (Hinck et al. 2006). High mortality rates in salmon can be attributed to lost riparian cover and temperatures exceeding 19°C (Gale et al. 2014; Jeffries et al. 2014), and high temperatures can also indirectly affect salmon populations by causing macroinvertebrates, an important food source, to mature faster but reach a smaller adult size (McCullough 2009). Thus, high water quality is critical for migratory fish species that need to use these stream reaches during their life cycle (Regetz 2003; Jorgensen et al. 2009).

Land use practices and/or conversions can negatively impact waterways, and even though land use conversion is necessary for social purposes, protection of water quality within stream systems is also an important societal goal for long-term sustainability of ecosystem services (Foley et al. 2005). The arid shrub-steppe within the valley contains critically important agricultural lands that require a large amount of irrigation water, but the streams support endangered species of salmon and other important ecosystem and cultural services, so water quality is an important management goal for diverse interest groups (Dittmer 2013; Macfarlane et al. 2017; Office of Columbia River 2018). Water quality is affected by many different variables unique to each watershed. Different land uses have point and non-point source pollution inputs that affect water quality differently. Poor water quality negatively affects biotic communities that depend on streams, as well as humans, who gain benefit from ecosystem services provided by streams with good water quality such as recreational opportunity, cultural values, and simply enjoying the aesthetic beauty of a river system. Determining how to improve water quality requires an in-depth understanding of how water quality is affected by spatial differences in land use. As such, it is important to study the associations between water quality and land use, as well as how water changes longitudinally through various land use types in Kittitas Valley.

Biophysical Study Area

Geology of the Kittitas valley is composed of layers of basalt millions of years old (Crawford C. 2003); over a million years ago, glacial ice cut into the basalt layers and deposited silt while rivers deposited alluvium in the valleys (Crawford C. 2003). Despite the relatively uniform geology across the study sites, important ecological differences exist. Level III ecoregions defined in the Kittitas Valley are the Columbia Plateau, Eastern Cascades Slopes and Foothills, and Cascades ecoregions (Omernik and Griffith 2010). Ecoregions are defined by similarities in biotic and abiotic factors within each landscape, and differences in climate, vegetation, geology, and hydrology can vary greatly between ecoregions (Omernik and Griffith 2010). The headwater reaches of the streams included in this study are all in the Cascades and Eastern Cascades level III ecoregions, and the downstream reaches are in the Columbia Plateau level III ecoregion (Omernik and Griffith 2010).

The rain shadow effect from the Cascade Mountains defines the climate of the study sites and influences the ecoregions (Siler et al. 2013). The rain shadow effect occurs when prevailing winds from the west cause greater amounts of precipitation on the windward side of and at the crest of the mountains compared to the leeward or east side of the Cascade mountain range (Siler et al. 2013). This causes big differences in rainfall and temperature which influence the ecology of the region, particularly moving from small, high elevation headwater streams to large, low elevation alluvial stream on the valley floors downstream from the Cascade crest. For example, the average temperature at 2200’ is 26°C in July compared to an average of 29°C at 1500’ (Figure 3). Moreover, average precipitation per year varies from 58.22 cm at 2200’ to 22.58 cm at 1700’ (Kittitas County 2020; Your Weather Service 2020).

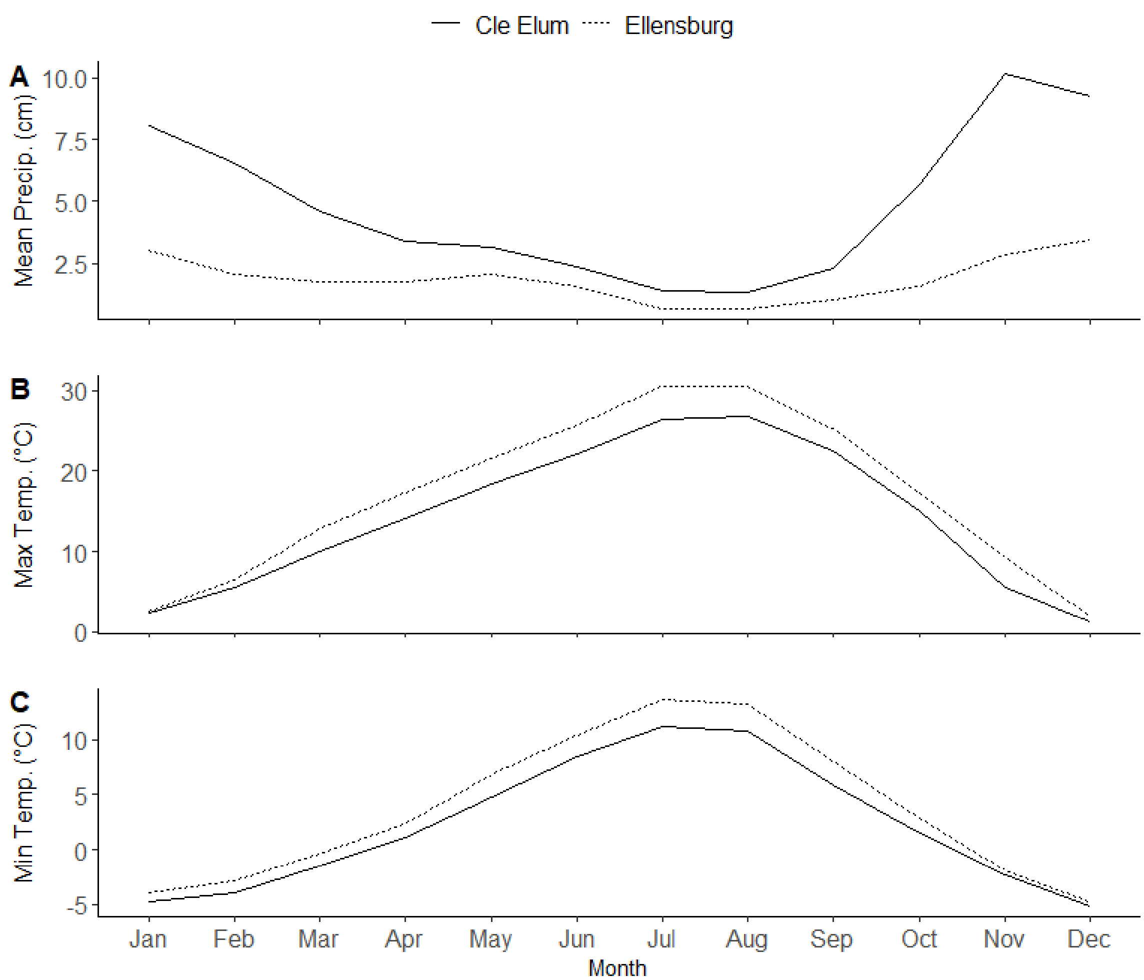

Figure 3 Variations between climate at lower and higher elevations. (A) Mean precipitation (cm) at 2200’ in Cle Elum, WA (solid line) and at 1700’ in Ellensburg, WA (dotted line). (B) Mean maximum temperature (°C) for Cle Elum and Ellensburg, WA. (C) Mean minimum temperature (°C) for Cle Elum and Ellensburg, WA (National Oceanic and Atmospheric Administration 2010).

The climates at different elevations could cause variation in stream temperature, macroinvertebrates, discharge, and dissolved oxygen among longitudinal samples taken with a given stream from upstream to downstream. The seasonal differences in precipitation are also important in shaping the hydrographs of streams in the valley. These streams naturally have a snowmelt hydrograph, but water stored in reservoirs is released during the summer for irrigation, which creates a consistent environment for sampling (Figure 4). The discharge pattern in the mainstem Yakima will be mimicked by my study streams feeding into the Yakima River due to irrigation delivery through the stream systems, except during hay cutting when irrigation is temporarily stopped (USGS 2018).

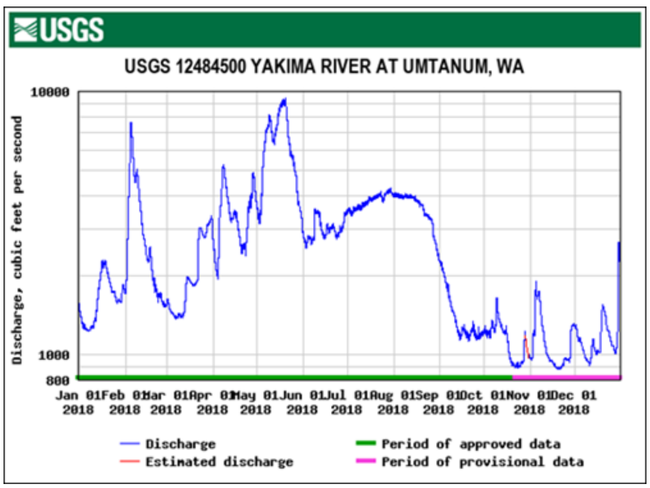

Figure 4 Hydrograph of the Yakima River near Umtanum creek confluence (USGS 2018). The steady flows from June through September correspond to irrigation delivery in the mainstem river.

The upstream forested areas of this study are dominated by coniferous pine forest of the Cascade foothills and Columbia Plateau ecoregion. The Columbia Plateau Ecoregion is characterized as shrub-steppe, which typically includes different sagebrush species, bitterbrush, and native and invasive grasses (Crawford C. 2003; Omernik and

Griffith 2010). As the rivers flow into the valley, the typical Columbia Plateau ecoregion vegetation transitions to willow-dominated (Family Salicaceae) riparian areas with large swaths of reed canary grass (*Phalaris arundinacea*) where the riparian vegetation has been disturbed. Other species found in riparian areas include alder (*Alnus rubra*), serviceberry (*Amelanchier alnifolia*), and Douglas maple (*Acer glabrum*) which grow where land use permits.

Cultural Study Area

This study takes place on lands that were managed by and which supported the Yakama Indian Nation. For thousands of years, the Yakama hunted animals and gathered food still considered culturally important today (Montag et al. 2014). In fact, all aspects of the stream system are culturally important to the Yakama Nation (The Yakama Nation 2016). For example, Pacific salmon are an important natural, economic, and cultural resource in the Pacific Northwest, and the Yakama Nation who have rights to the resources in their ceded lands, including the Kittitas Valley, value healthy stocks of salmon that spawn in these rivers (Fears 2015; The Yakama Nation 2016). The streams that support migratory fish must meet certain water quality conditions for them to successfully spawn and rear, including habitat for insects that juvenile fish eat, cool temperatures during summer, and appropriately sized substrate types for spawning (Jorgensen et al. 2009).

Today the Yakama Nation works with agencies like Kittitas County Conservation District (KCCD) and Washington State Department of Ecology (DOE) that have different water quality monitoring and restoration programs for Washington state. For example, DOE has a watershed monitoring program to comply with the federal Clean Water Act (WA State Department of Ecology 2019), and recently KCCD opened miles of spawning habitat in Manastash Creek (Kittitas County Conservation District). KCCD also has programs to encourage landowner compliance with fish passage and screening laws for irrigation canals (Kittitas County Conservation District). As an example of other restoration efforts by KCCD, a levee was recently removed on Reecer Creek, which was also re-meandered/lengthened. These activities demonstrate the local demand for improving stream habitat for fisheries and other ecosystem services, often in collaboration with the Yakama Nation

Water quality in Kittitas Valley also provides an economic value to the people living here (Montag et al. 2014; Office of Columbia River 2018; The Yakama Nation 2019). With water-dependent economic output over $13 billion dollars in 2018 and ranked third for water dependent employment in the state, the Yakima River Basin is economically important to our region (Office of Columbia River 2018). Agricultural land use dominates the lowlands in the study area, consisting mostly of hay, alfalfa, and cattle. In 2012 total farmland in Kittitas County was 183,124 acres with 1,006 farms (USDA 2012) that generated almost $69 million dollars (USDA 2012). Farmers rely on high quality water to irrigate crops and water cattle.

Another major use of the county today is recreation, including fishing, hiking, horseback riding, biking, and winter sports. This requires high water quality to maintain habitat. The recreation value of a clean and healthy river gives people the opportunity to fish, swim, or float the Yakima River. This recreational value includes the scenic value of the stream systems and riparian areas within this valley, where well-used walking trails line rivers so people can view native birds and other wildlife attracted to the water. Excess pollutants such as elevated levels of suspended sediment can decrease recreational value because people do not like swimming in streams that are not clear.

While collecting data for this thesis, I was struck by the number of people I saw who clearly valued the stream systems flowing through the valley. For example, I saw people with their children swimming in the river, one farmer asked about my project and why I was in the stream, and another homeowner along Cooke Creek stopped and asked me questions about stream biology and water quality. All these interactions show that people truly care about the streams in their environment as well as the ability to enjoy the ecosystem services they provide.

CHAPTER III

METHODS

Study Design

All streams chosen for this research flow through at least two of the three land uses being studied: forested, agricultural, and urban. In this study, forested land use is defined as public forest, commercial forest, and/or open space land, as defined by RCW 84.34.020; agricultural land use is defined as classified and non-classified agricultural areas; and urban land use is defined as single and multi-family homes, parking lots, industrial, and retail land use (Washington State Legislature 2014). Six streams were selected for this study (Figure 5): Reecer, Wilson, Naneum, Coleman, Cooke, and Umtanum. From within these six streams, thirteen sample sites were selected: five forest land uses, five agriculture, and three urban sites (Figure 5). Umtanum, which is not in the Kittitas Valley, was chosen to represent a lower elevation forested site. Although much of Umtanum is not literally forested, it is in the "forested" land use classification because it is largely undeveloped. This allowed for a large enough sample size to draw conclusions about how land use effects water quality.

Sampling was done three different times during summer 2019. The first sampling period was from July 14th through July 27th, the second from August 11th through August 24th, and the third from September 8th through September 21st. Sampling was done during the summer to include the effects of irrigation return flow on the stream system. Replication through time allowed variation in irrigation delivery to be assessed.

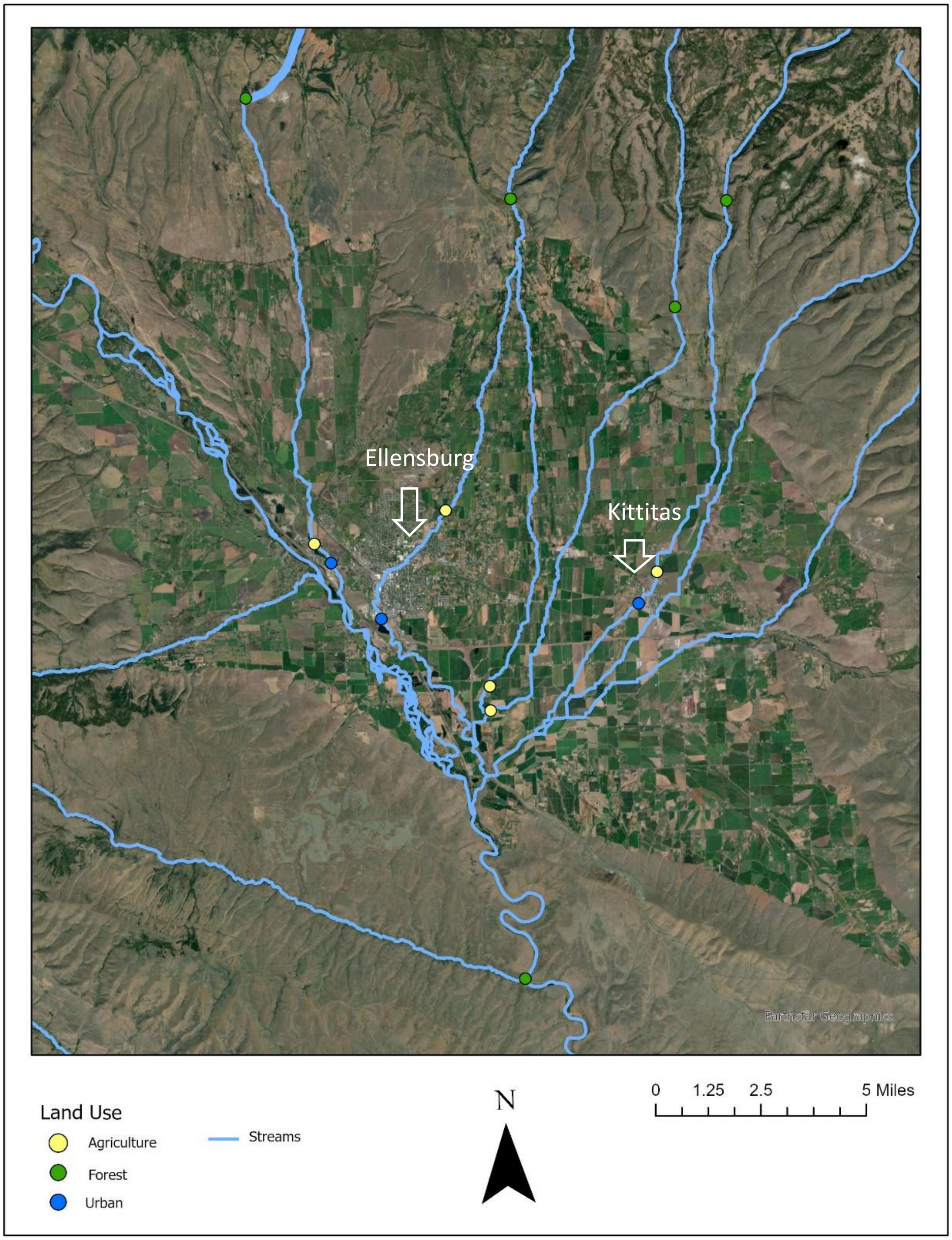

Figure 5 Map of the study sites in and near the Kittitas Valley (ESRI 2018).

At each sample location, I completed a site description that included Proper Functioning Condition Assessment for lotic ecosystems (US Department of Interior 2003), GPS coordinates and elevation (My Elevation android app by RDH Software, version 1.59, 2014), Wolman pebble count (Wolman 1954), thalweg depth, and bank full width (Table 1). Thalweg was determined by measuring the depth at the representative cross section and determining the deepest part of that section. Bank full width was determined by using that same representative cross section and visually determining were the stream is at bank full then measuring the width. These variables were measured to give an initial description of the site conditions both in the stream and in the immediate riparian areas.

Sampling Strategy

All samples were taken as far downstream of the targeted land use as possible to maximize the percentage of that land use in the watershed. All samples were taken between 11 AM and 3 PM to minimize differences among sites and sample date due to time of day.

Depth and Discharge

I chose a representative cross section of the stream at each sample site that was marked with GPS. The same spot was used to measure depth and discharge for all three sampling periods. Depth was measured every half meter with a meter stick, then averaged. Discharge was measured with a Swoffer flow meter. Velocity measurements were taken every half meter at 60 % depth for 5 seconds. Discharge was calculated with the following equation:

$$\text{Discharge} = \text{width} \times \text{depth} \times \text{water velocity}$$

Table 1 Physical site characteristics. Proper Functioning Condition (PFC) rating: PFC=Proper Functioning Condition, FAR=Functional at Risk, NA=Not Apparent.

Stream	Land Use	Elevation (m)	Pebble Mean (mm)	Pebble Median (mm)	Bank full (m)	Thalweg Depth (m)	PFC Score	PFC Score Trend
Reecer	forest	2806	27.5	25.2	0.7	0.1	PFC	Down
Reecer	ag	1537	<2.0	<2.0	5.0	1.0	FAR	NA
Reecer	urban	1525	24.2	22.5	5.7	0.6	PFC	Up
Naneum	forest	2455	94.1	89.0	7.5	0.5	PFC	NA
Naneum	ag	1483	30.8	30.3	3.2	0.3	FAR	Down
Wilson	ag	1687	54.9	55.4	3.2	0.2	PFC	NA
Wilson	urban	1494	15.8	15.2	5.3	0.5	FAR	NA
Coleman	forest	2388	56.6	51.2	3.5	0.2	PFC	Down
Coleman	ag	1461	37.2	35.1	2.9	0.8	FAR	Down
Cooke	forest	2915	87.2	86.1	2.4	0.2	PFC	Down
Cooke	ag	1699	63.4	58.8	2.0	0.1	FAR	Down
Cooke	urban	1637	54.9	49.9	2.1	0.4	FAR	Down
Umtanum	forest	1345	58.6	50.2	1.3	0.1	PFC	Down

Suspended Sediment

To obtain total suspended sediment, a DH-48 suspended sediment sampler was used to collect approximately 500 mL of stream water. The sampler was moved slowly up and down in the thalweg of the stream until the bottle was full. Samples were measured in a 500-mL graduated cylinder to obtain exact volume before filtering through an ashed 1 µm glass fiber filter. The filtered sediment was dried for 24 hours at 65 °C, then weighed. The filters were then ashed at 500 °C for six hours. The initial filter weight was subtracted from the un-ashed and ashed filter weight, then the ashed filter weight (inorganic suspended sediment) was subtracted from the un-ashed filter weight to obtain the mass of organic suspended sediment, which was expressed as g/L.

Turbidity

Turbidity was obtained by collecting approximately 100 mL of water in a clean glass jar at 60 % water depth in the thalweg of the stream. This sample was taken to the lab, and turbidity was measured on a calibrated Orbeco-Hellige model 966 turbidimeter. Each sample was turned three times before taking the measurement to ensure uniform distribution of particles.

Specific Conductivity, Temperature, Dissolved Oxygen, pH

A Yellow Springs Instruments (YSI) model 85 DO and conductivity meter was used to measure specific conductivity, temperature, and dissolved oxygen at the sites. Before each measurement, the instrument was calibrated to the elevation of each site. Elevation was estimated using a phone application called My Elevation (version 1.60, RDH Software, 2014). The probe was held in the thalweg at 60 % water depth, and the measurements were recorded three times each and averaged. An ISFET pH meter (model IQ120) was used to measure pH at each of the sites in the thalweg of the stream at 60 % water depth.

Ammonium, Nitrate, and Phosphate

Water samples were collected in acid washed high density polyethylene (HDPE) bottles rinsed with filtered site water to measure ammonium, nitrate, and phosphate concentrations. The stream water samples were filtered through a glass fiber filter with 1 μm nominal pore size, stored on ice, and frozen upon return to the laboratory within 24 h. Ammonium was measured using the fluorometric method (Holmes et al. 1999; Taylor et al. 2007) on a Turner Designs benchtop fluorimeter. Nitrate was measured using the cadmium reduction method (U.S. Environmental Protection Agency 1983; APHA 1992) which also measures nitrite, but because nitrite values are often negligible, I hereafter refer to these measurements as just nitrate. Phosphate was measured as soluble reactive phosphorus using the ascorbic acid method (Murphy and Riley 1962; Edwards et al. 1965; APHA 1992). Nitrate and phosphate were both measured on a SEAL AQ1 discrete analyzer.

Macroinvertebrates

A D-frame kick net with a mesh size of 500 μm was used to sample benthic macroinvertebrates. The net was placed on the bottom of the stream and the gravel upstream was kicked for 10 seconds. Insects that washed into the net were then turned out into a collection tray. This procedure was done in all microhabitats of the stream, including banks, the center of the stream, pools, riffles, backwater outlets, areas with large amounts of leaves, and any other unique feature of each stream sites. Macroinvertebrate samples were immediately stored in 95% ethanol for transport back to the lab and at a later date, they were identified to family level according to Merritt, Cummins, and Berg (2008). Composition of the benthic community was analyzed using the EPT index. The EPT index is typically a species level identification that compares the percentage of Ephemeroptera, Plecoptera, and Trichoptera larvae to the rest of the larvae in the sample. High EPT percentages indicate better water quality (US EPA). Although most researchers use this index at species level, there is evidence that family level identification is sufficient (Watershed Science Institute; Herman and Nejadhashemi 2015).

I also used the Hilsenhoff Family Biotic Index (HBI) as an alternate measure of insect community condition which assigns tolerance values for each macroinvertebrate family. This is another aquatic macroinvertebrate biotic index used to infer water quality patterns, and it only requires family level identification. The number of insects in a particular family is multiplied by the tolerance value, summed, and divided by the product of total of counts and tolerance values. The smaller the number produced, the better the water quality.

Percentage Land Use

Using the buffer function (ArcGIS Pro version 2.6.1), a 100-meter buffer on the stream corridor was created to calculate percentage land use upstream of each site. This buffer evaluation was calculated from the sample location upstream to the next sample location. For forested sites, the 100-meter buffer was calculated from the sample location to the end of the stream based on the stream data from DOE. The streams layer came from the Kittitas County GIS data webpage and was corrected through ESRI imagery (ESRI 2018). Land use data was from the 2010 statewide land use data from Washington State Department of Ecology (Washington State Department of Ecology 2010). This was also updated visually through ESRI imagery to account for land use changes between 2010 and 2019 when the samples were taken. For example, roads were not included in this land use data and had to be added to urban land use. Also, this land use data was not accurate enough for the detailed analysis that I needed, so I went through and more accurately outlined the land uses within the buffer chosen. Through visual analysis, areas that had agricultural land use, for example, that overlapped a road or house slightly were fixed to be more accurate with visual observations.

Statistical Analysis

General linear models were used to examine how land use affected the water quality response variables. Analyses included stream as a fixed effect instead of a random effect, allowing management conclusions to be drawn about these specific streams, and an interaction between land use and month was a main effect. For turbidity, patterns in the data indicated that an interaction between land use and stream should also be used in addition to an interaction between land use and month.

For all analyses, the residuals of each model were plotted and analyzed with Shapiro-Wilks Normality test to ensure that model assumptions were met, and if not, data were transformed (Table 2). A likelihood ratio test was performed on the model made for each response variable and a model that only differed by omission of the land use factor, resulting in a chi-square value. If the likelihood ratio test finds a significant difference between the full model and the model without land use, it indicates that land use was a significant influence on the response variable. To further examine the interaction between land use and the response variables, a principal component analysis (PCA) was performed that included land use percentages within the 100-meter buffer upstream of each site. All statistical analysis was done in R statistical software (R Core Team 2013), and statistical significance was determined at α of 0.05. A summary of the process for analyzing these data is shown in Figure 6.

Table 2 Statistical transformation, interaction, and normality test for each response variable.

Variable	Transformation	Interaction in Model	Shapiro-Wilk Normality Test	p-value
HBI	N/A	Yes	0.990	0.963
EPT	N/A	Yes	0.961	0.160
Nitrate	Log	Yes	0.984	0.798
Ammonium	N/A	No	0.982	0.729
Phosphate	Log	Yes	0.949	0.062
pH	N/A	Yes	0.972	0.383
Temperature	N/A	Yes	0.966	0.238
Turbidity	Log	Yes	0.978	0.579
Dissolved Oxygen (mg/L)	N/A	Yes	0.951	0.072
Dissolved Oxygen %	Log	Yes	0.977	0.555
Inorganic Suspended Sediment	Log	No	0.976	0.516
Organic Suspended Sediment	Square Root	Yes	0.971	0.356
Specific Conductivity	N/A	Yes	0.961	0.162
Average Depth	Log	Yes	0.979	0.635

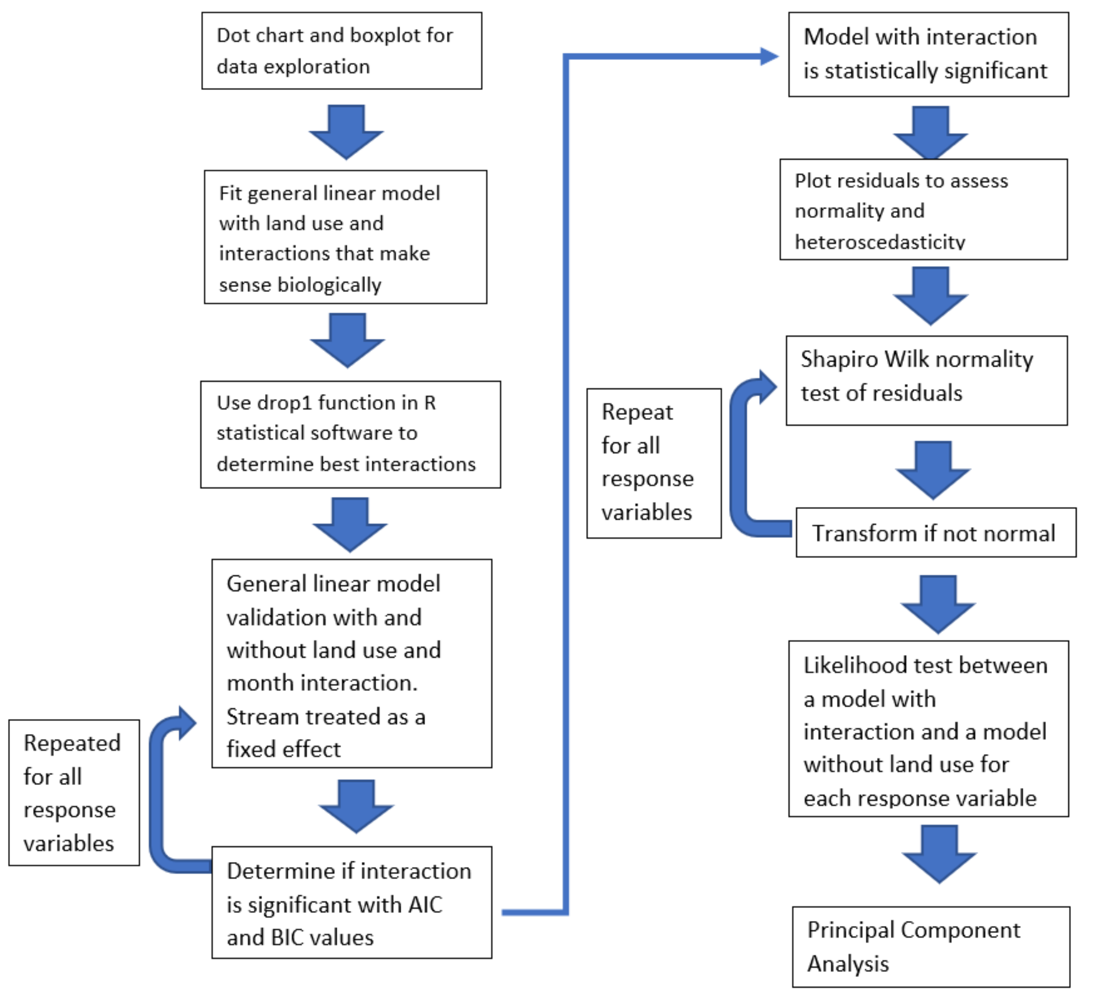

Figure 6 Flow chart of statistical analysis and modeling of the independent and response variables.

CHAPTER IV

RESULTS

Land Use Percentage

I used an evaluation of the land use in a 100-meter buffer of each stream to quantify the percentage of each land use affecting each site. Cooke forest was the only site with 100% forested land use with a mix of private and public forests whereas Coleman forest had 88.7% forest. Reecer forest had 49.5% agricultural land use upstream of the site and 42% forested land use (Figure 7) due to the classification of livestock grazing (Washington State Legislature 2014). Reecer urban had 73.7% agricultural land use and 25% urban land use (Figure 7). This graph does not differentiate between private/commercial timber land and other forest types (open space, public land, parks), which could have an effect on water quality in the forested sites.

Discharge, Average Depth, Suspended Sediment

In all sample periods, the forested sites had significantly less overall discharge which also remained consistent among the three sampling periods. In July, average discharge of agricultural sites matched the forested sites, but then increased in August and again in September (Figure 8). Forested sites ranged from 5 L/s to 434 L/s, agricultural sites ranged from 27 L/s to 838 L/s, and urban sites ranged from 166 L/s to 856 L/s. Average depth of the sampling sites had a significant interaction with land use. In this interaction, forested sites remained less than 0.25 m depth through the study whereas urban sites were deepest in the August sampling period. Moreover, the depth of forested and urban sites varied less through each sampling period compared to agricultural sites (Figure 8). The average depth for agricultural sites was 0.46 m while urban sites had an average depth of 0.41 m among all sites and sample times (Figure 8). There were no significant differences in inorganic or organic suspended sediment among forest, agricultural, and urban land use, or sample times (Figure 9).

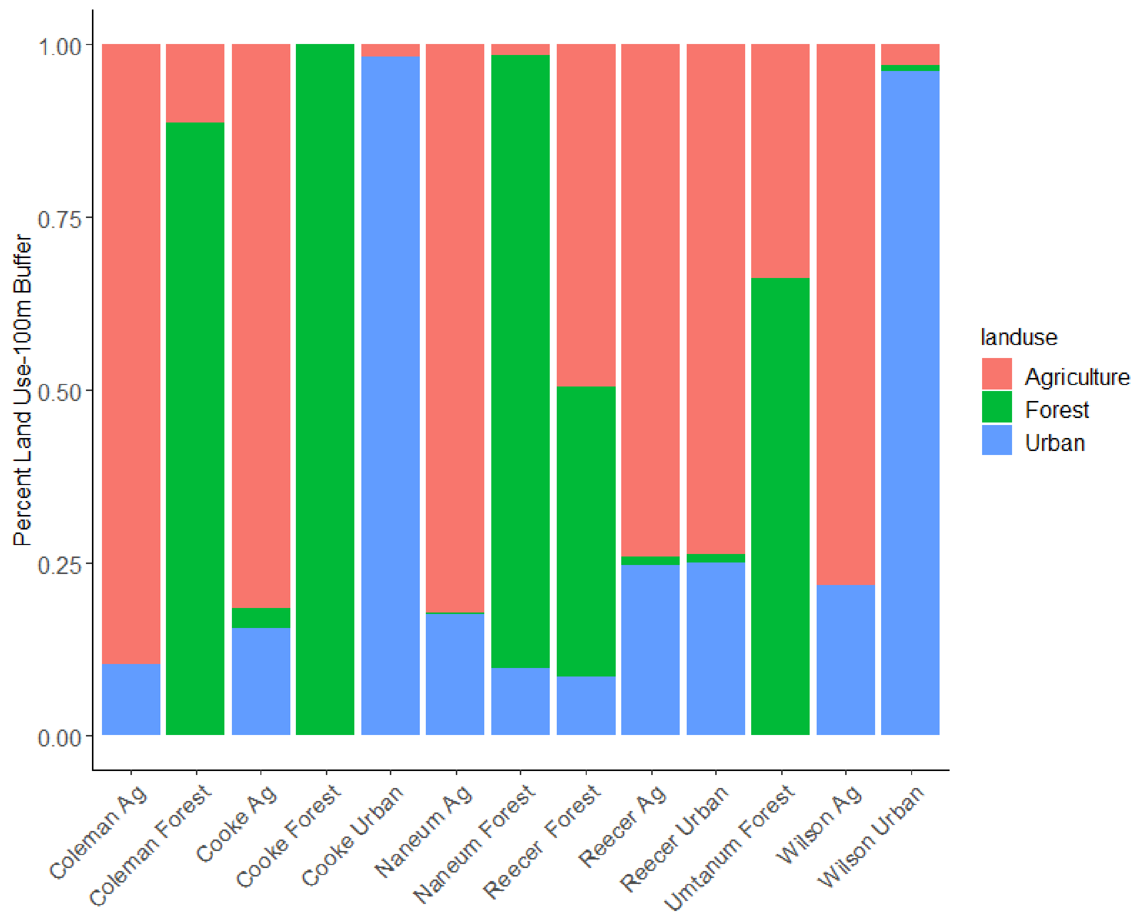

Figure 7 Percentage of land use upstream of each site within a 100-meter buffer of each stream, calculated by using entire length of stream upstream of given sample location.

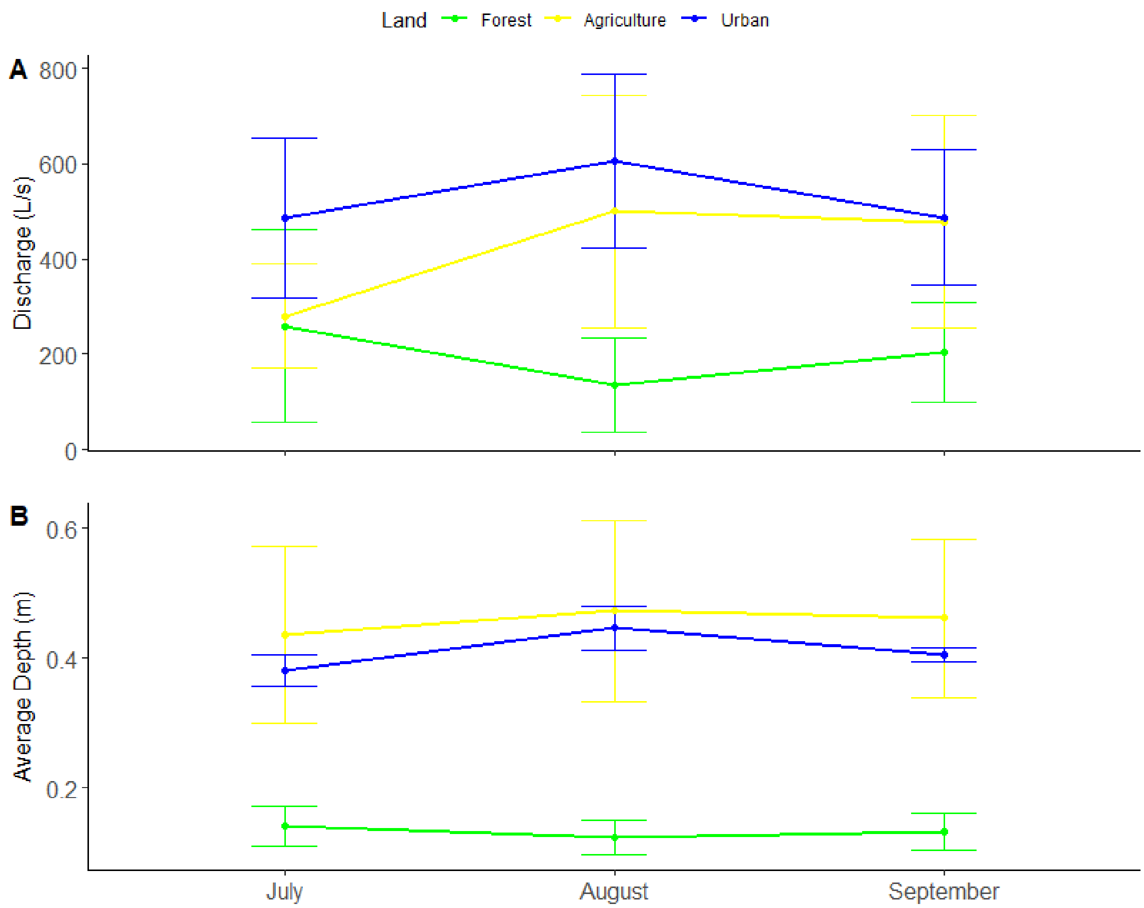

Figure 8 Average data for each month separated by land use. (A) Discharge (L/s) varies among land uses (Chisq=25.287, df=6, p-value<0.001). (B) Average depth (m) interacts with land use (Chisq=26.947, df=6, p-value<0.001). Error bars represent 1 standard error.

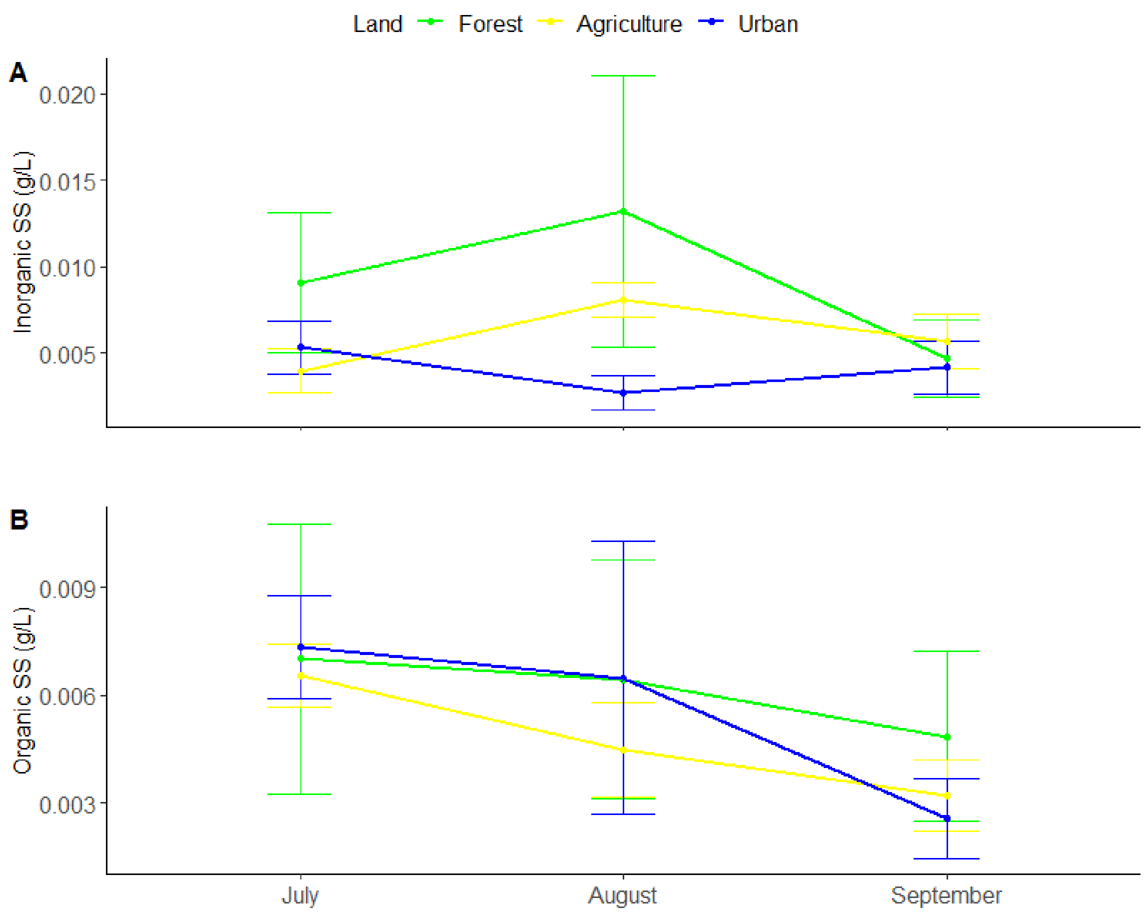

Figure 9 Average data for each month separated by land use. (A) Inorganic suspended sediment (g/L) does not interact with land use (Chisq=2.7134, df=2, p-value=0.258). (B) Organic suspended sediment (g/L) does not interact with land use (Chisq=2.7697, df=6, p-value=0.837). Error bars represent 1 standard error.

Specific Conductivity and Turbidity

Specific conductivity and land use significantly interacted. Specific conductivity of forested sites stayed at around 100 µS per cm among sample periods (Figure 10) but ranged from 175 to 200 µS per cm in agriculture and urban sites (Figure 10). Turbidity did not interact with land use, and samples from forested sites varied more widely than other land uses (Figure 10).

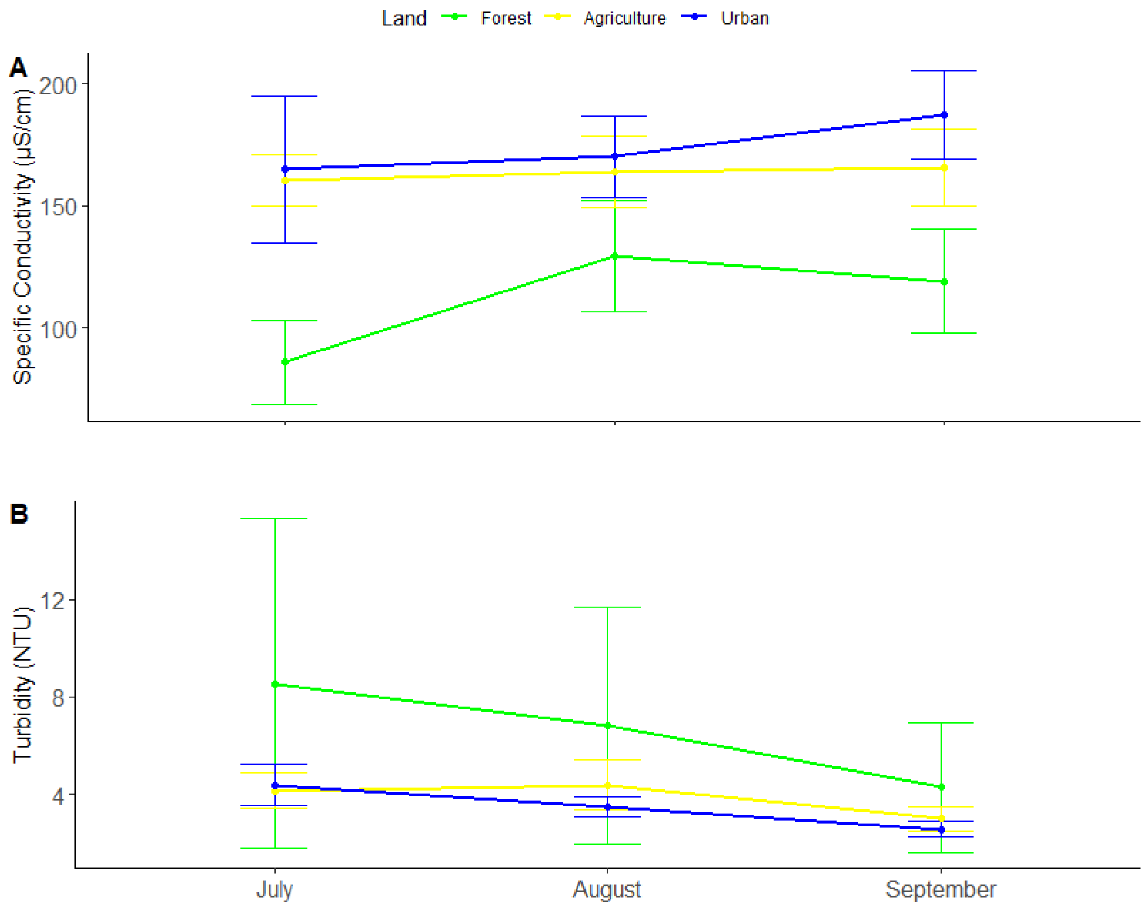

Figure 10 Average data for each month separated by land use. (A) Specific conductivity (µS/cm) interacted with land use (Chisq=28.381, df=6, p-value<0.001). (B) Turbidity (NTU) did not interact with land use (Chisq=3.8474, df=4, p-value=0.427). Error bars represent 1 standard error.

Dissolved Oxygen

Urban sites had significantly lower dissolved oxygen (mg/L) than forested and agricultural sites (Figure 11). Dissolved oxygen averaged around 8.50 (mg/L) in urban sites at all time periods, while dissolved oxygen in forested sites ranged from 9.49

(mg/L) to 9.72 (mg/L) (Figure 11). Percentage dissolved oxygen did not interact with land use, so there were no differences among sites (Figure 11).

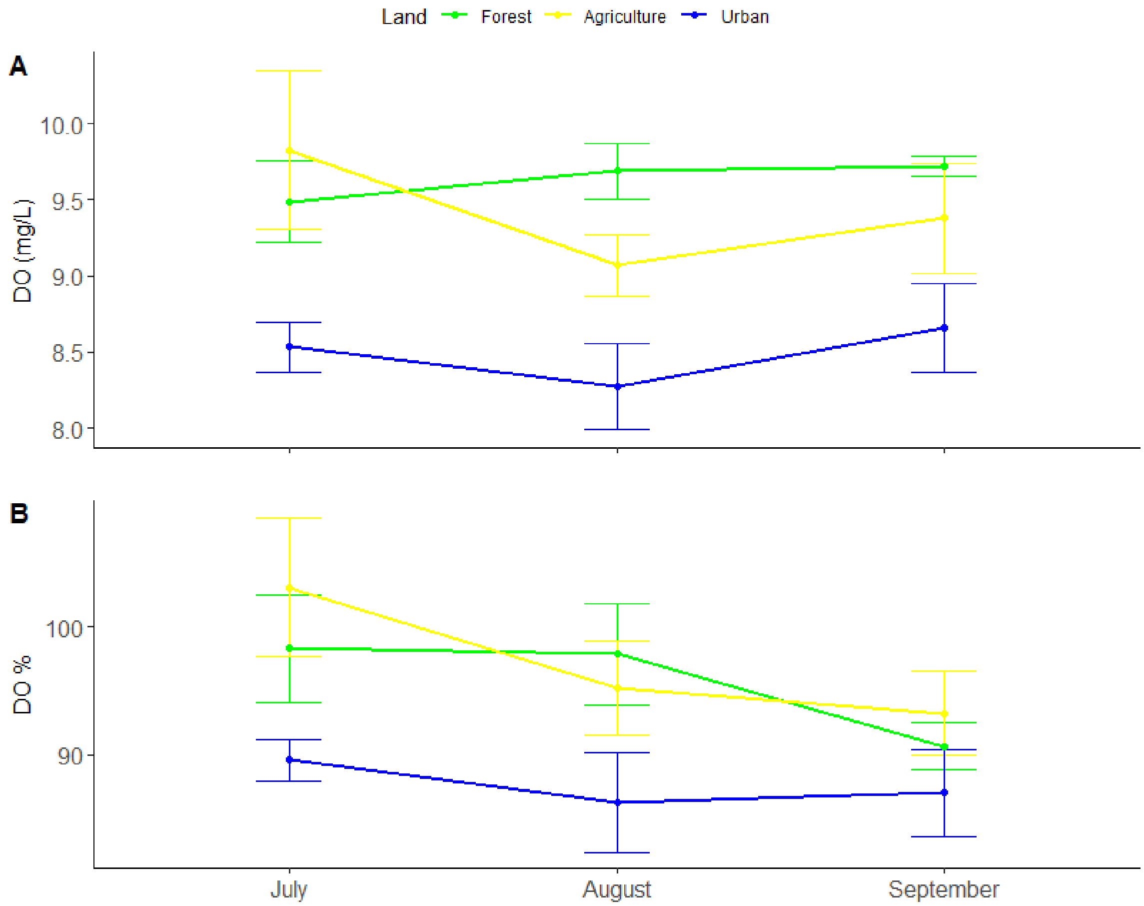

Figure 11 Average data for each month separated by land use. (A) Dissolved oxygen (mg/L) interacts with land use (Chisq=16.958, df=6, p-value=0.009). (B) Dissolved oxygen (%) does not interact with land use (Chisq=7.9638, df=6, p-value=0.241). Error bars represent 1 standard error.

Temperature and pH

Temperature varied significantly by land use. Forested sites were much cooler, reaching a high of 14°C in August, then dropping again in September to 12°C (Figure 12). Temperatures in agricultural and urban sites remained higher than forested sites throughout this study, never dropping below 13°C. There was no significant difference in pH among the study sites, which varied between 7.0 and 8.0 with a few streams being close to 8.5 or 6.5 in September (Figure 12).

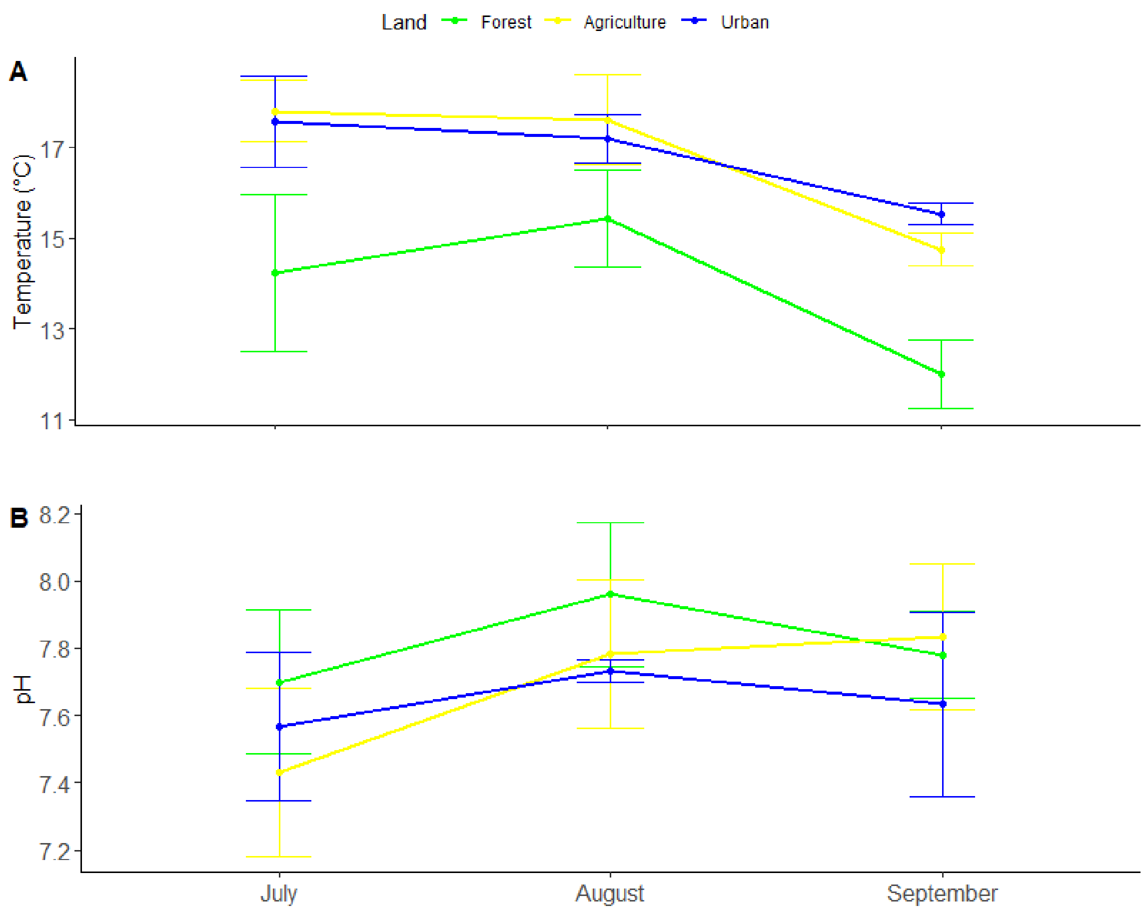

Figure 12 Average data for each month separated by land use. (A) Mean temperature (°C) interacts with land use (Chisq=53.803, df=6, p-value<0.001). (B) Mean pH does not interact with land use (Chisq=4.1002, df=6, p-value=0.6631). Error bars represent 1 standard error.

Ammonium, Nitrate, and Phosphate

Ammonium (µg/L) concentrations did not vary by land use, but they varied more widely within agricultural sites in July compared to urban and forested sites. In contrast, ammonium concentrations from August and September were very similar among land uses (Figure 13).

Nitrate concentrations (mg/L) significantly differed among land uses with forested sites having average nitrate concentrations close to zero throughout the study (Figure 13). Nitrate in urban and agricultural land uses was high in July, averaging 0.27 mg/L and 0.32 mg/L respectively, but it dropped to an average of 0.16 mg/L in urban areas and 0.14 mg/L in agricultural areas in August (Figure 13). Nitrate concentrations were highly variable for urban sites in September but had the highest average concentration between all land uses and months at 0.35 mg/L (Figure 13).

Phosphate concentrations significantly differed by land use. Concentrations of phosphate in forested sites averaged 0.033 mg/L in July and 0.034 mg/L in August and September (Figure 13). Agricultural sites averaged 0.057 mg/L in July then dropped to an average of 0.039 mg/L in September. Urban sites varied in phosphate concentrations throughout the study but remained higher than other land uses (Figure 13).

EPT and HBI

EPT had a significant interaction with land use, and average EPT percentage was higher in forested sites compared to agricultural or urban sites (Figure 14). Average EPT percentages were highest in July at 63.2% in the forested sites while urban sites were 43.6% and agricultural sites were 44.6%. Throughout the study EPT percentages in forested sites stayed consistent, but in September, urban sites fell to 32.9% and agricultural sites fell to 33.3%.

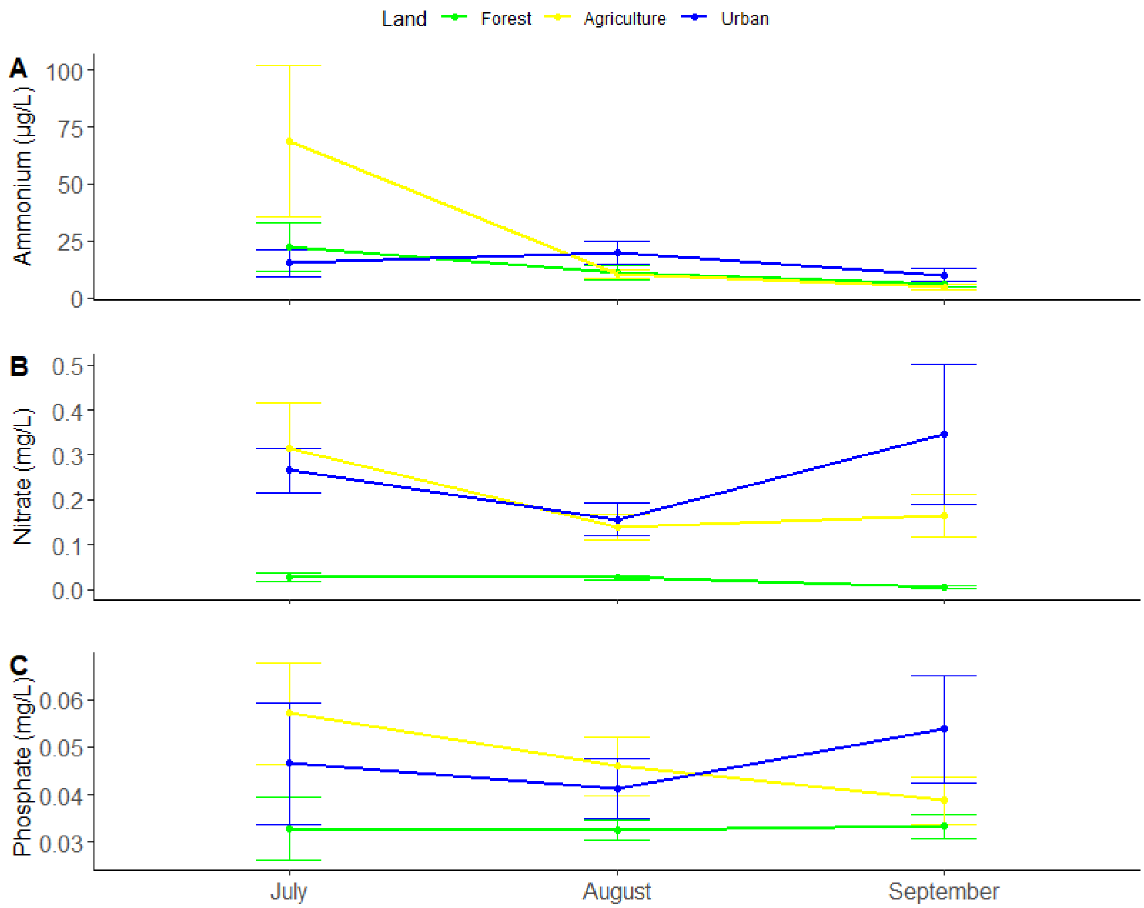

Figure 13 Average data for each month separated by land use. (A) Ammonium (µg/L) did not interact with land use (forest, agriculture, and urban) (Chisq=2.3639, df=2, p-value=0.3067). (B) Nitrate (mg/L) varied among land use (Chisq=56.756, df=6, p-value<0.001). (C) Phosphate (mg/L) varied by land use (Chisq=32.219, df=6, p-value<0.001). Error bars represent 1 standard error.

HBI uses tolerance values of insects and indicates better water quality (i.e., more sensitive insect families) with lower numbers. HBI had a significant interaction with land use (Figure 14). Forested sites had scores between 0 and 3.75 in all months, the lowest of all three land uses, indicating a rating of “Excellent” water quality (Hilsenhoff 1988). Narrow standard errors of the mean in the forested sites indicate little variance among the different streams. The average HBI in agricultural sites had a rating of “Good” water quality, and the scores stayed between 4.26 and 5.00. Average HBI for urban sites in this study were similar to agricultural sites but did get some scores of “Fair” and “Fairly Poor” water quality. Urban scores were between 4.26 and 6.50. HBI increased at urban and forested sites during the study but stayed consistent in agricultural sites (Figure 14). Because forest sites are upstream from agricultural sites, and agricultural sites are upstream of urban sites, this shows a pattern of steadily degrading water quality as streams flow through the Kittitas Valley.

Principal Component Analysis

Principal component analysis was used to find broad patterns in the aggregated data (Table 3). Six principal components explained 82% of the variation in the data. The PCA revealed that streams with large proportions of agricultural land use in the 100-meter buffer had a high positive correlation with phosphate and a weak positive correlation with ammonium (Figure 15). Agricultural land use was also significantly correlated with higher depth and increased nitrate. Urban land use was highly correlated with agricultural land use, also having high correlations with nitrates, phosphates, and high average stream depth. In addition, urban land use was highly correlated with higher stream temperature and specific conductivity, and insects with higher tolerance values (HBI). Urban and agricultural land use were also correlated with faster discharge (Figure 15). Forested land use was highly correlated with a high EPT value, and forested land use was distinctly uncorrelated with all other variables (Figure

15). Turbidity was highly correlated with inorganic and organic suspended sediment and was negatively correlated with dissolved oxygen and pH (Figure 15).

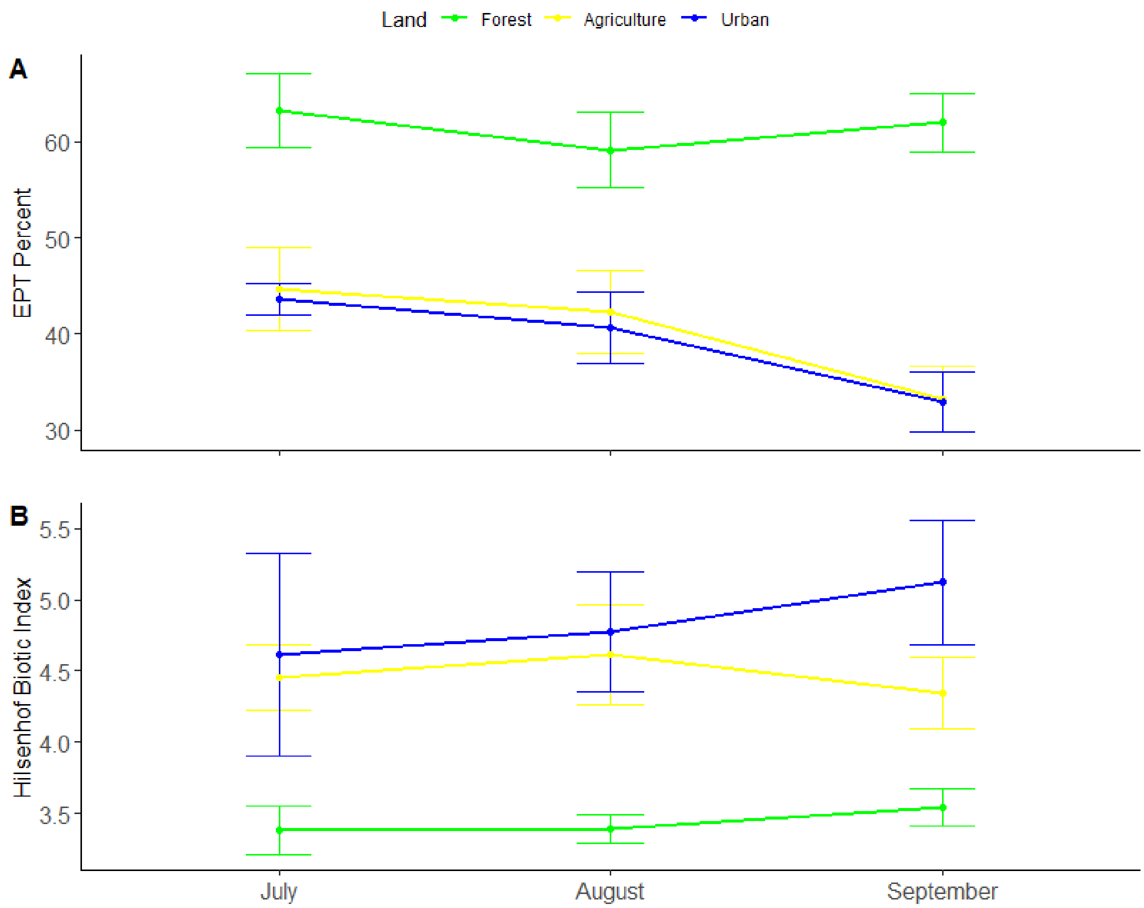

Figure 14 Average data for each month separated by land use. (A) EPT percentage is significantly affected by land use (Chisq=53.48, df=6, p-value<0.001). (B) HBI is significantly affected by land use (Chisq=25.466, df=6, p-value<0.001). Error bars represent 1 standard error.

Table 3 PCA correlation components and their importance in the analysis.

Principal Component Analysis-Importance of Components						
Component	1	2	3	4	5	6
Standard Deviation	2.340	1.675	1.626	1.399	1.227	1.106
Proportion of Variance	0.288	0.148	0.139	0.103	0.079	0.064
Cumulative Proportion	0.288	0.436	0.575	0.678	0.757	0.821
Eigen Values	5.477	2.806	2.642	1.958	1.506	1.223

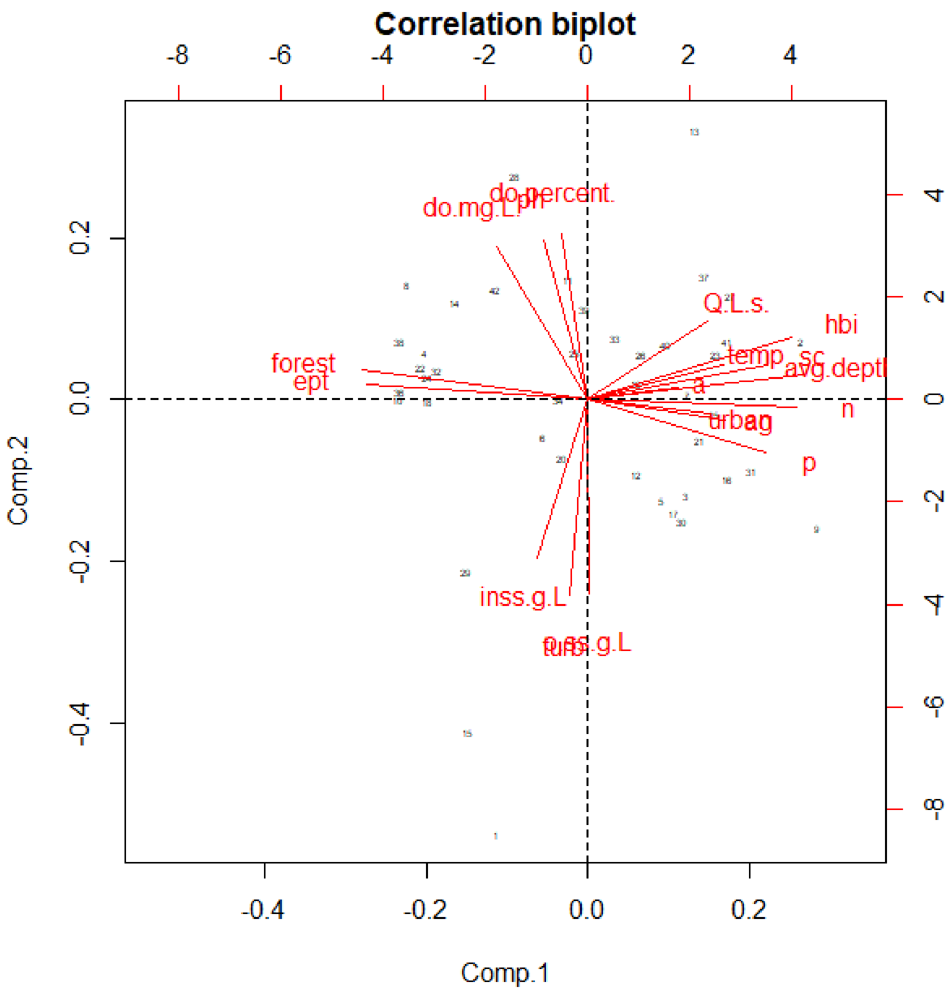

Figure 15 Correlation biplot of first two PCA components. Lines show the loading of each variable in this study. The longer the line, the better the correlation of the variables. Lines that are long and opposite from each other have strong negative correlations.

CHAPTER V

DISCUSSION

The primary goal of this study was to explore the relationship between land use (forest, urban, agriculture) and water quality and stream characteristics in the Kittitas Valley. I found that urban and agricultural land use was associated with increased stream depth and discharge compared to forested land use. There was no evidence that land use affected suspended sediment or turbidity, but forested sites had cooler stream temperature during the July through September sampling period. Although dissolved oxygen saturation was not related to land use, the amount of dissolved oxygen (mg/L) was higher in forested sites throughout the study. Unlike ammonium concentration which was not influenced by land use, nitrate and phosphate had higher concentrations in urban and agricultural sites. Both EPT and HBI showed more pollution intolerant families in forested sites and the disappearance of those families in urban and agricultural sites, which favored pollution tolerant families.

I found that agricultural and urban streams were associated with more discharge and deeper channels than forested streams. To shorten the duration of flooding in urban and agricultural areas and to hasten runoff from the landscape, streams throughout the US have been modified by channelization and straightening (Kuenzler et al. 1977), which was readily apparent in my agricultural and urban sites (Figure 16). Channelization straightens the stream and together with embankment alteration makes it deeper, allowing the stream to hold more flood water (Rambaud et al. 2009). These types of modifications have been shown to sometimes have the unintended effect of slowing down the stream and allowing sediment to drop out of the water column to increase fine sediment (Rambaud et al. 2009). Channelization has also been shown to decrease the number of riffles and pools within the stream, decreasing habitat (Rambaud et al. 2009). The substrate in agricultural and urban streams I studied was composed of mostly sand with little heterogeneity in particle sizes, consistent with previous studies (Negishi et al. 2002; Rambaud et al. 2009; Pedersen et al. 2014). I frequently observed sand surrounding larger sized substrate in the agricultural and urban streams with mean pebble counts between <2 mm and 63.4 mm (Figure 16) whereas substrate composition in forested sites was much more diverse and had higher Wolman pebble counts ranging from an average of 27 mm to 94.1 mm (Figure 16). Like many streams impacted by land use, my study sites showed signs of channel deepening and associated substrate simplification consistent with other degraded streams (Paul and Meyer 2001; Walsh et al. 2005; Gordon et al. 2007). The simplified substrate in these channelized streams is consistent with decreased biological function because of habitat simplification.

Although many studies show higher total suspended sediment and turbidity due to agricultural and urban land use (Waters 1997; Buck et al. 2004; Walsh et al. 2005; McDowell et al. 2017), I did not find a land use effect. This could be due to forested sites having some form of agricultural land use including logging or grazing (Figure 17A), which can be sources of suspended sediment in streams (Waters 1997; Cassiano et al. 2020).

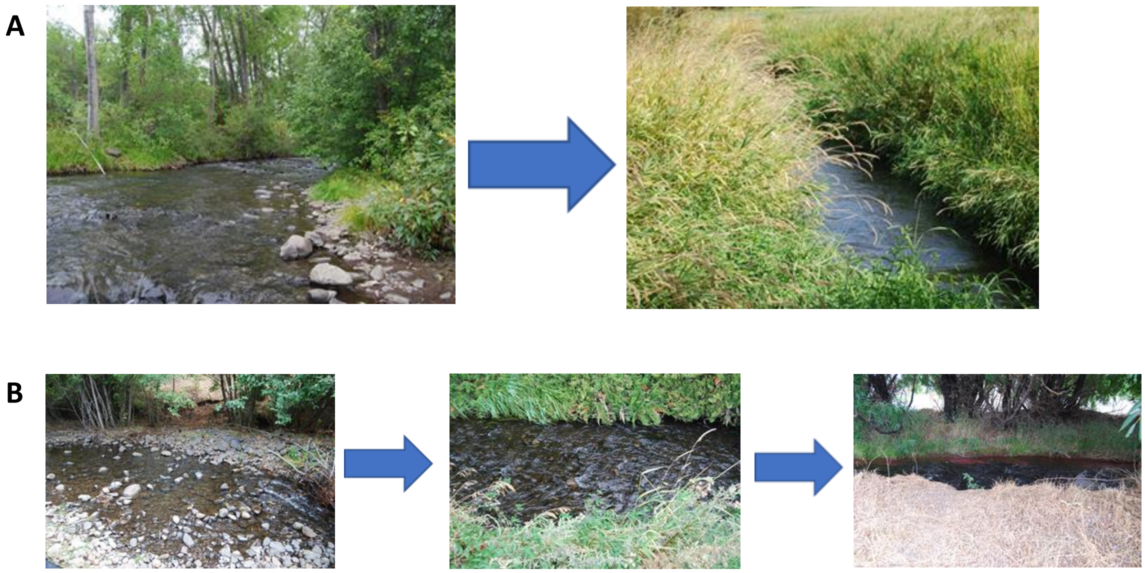

Figure 16 Visual evidence of stream channelization. (A) Naneum stream from the forested to the agricultural sample site. (B) Cooke stream from forested to agriculture to urban sample site.

Cassiano et al. (2020) found that streams in working forests that have been 15 % logged have 1.6 mg/L of total suspended sediments (TSS) and a forest logged at 88 % to have 6.1 mg/L of TSS, while in my study forested sites ranged from 3.7 mg/L to 62.4 mg/L of TSS, consistent with relatively high TSS in forested streams. Alternatively, the dominance of hay and pasture in agricultural sites and the general lack of urban runoff in the dry summer could cause the land use impacted sites I studied to have less turbidity than those in other studies of agricultural or urban streams. Because runoff from storms or snowmelt generally increase suspended sediment, it is possible that sampling in other seasons would yield different results that would show a difference among land uses.

Forested land use was associated with lower specific conductivity compared to agricultural and urban land use, indicating a lower number of solutes in the stream. This is consistent with other studies that have found that urban land use or increased impervious surfaces can increase conductivity (Dow and Zampella 2000; Conway 2007; Tu 2013). Another study found that the number of people with an area of developed land use, instead of the percentage of specific land use in a watershed, had a larger impact on specific conductivity (Tu et al. 2007). In my study, I found that specific conductivity increased with urban and agricultural land use equally, consistent with other previous studies (Morgan and Good 1988; Zampella 1994; Shupe 2017). Increased specific conductance has been associated in previous studies with lower biotic integrity in streams (Kimmel and Argent 2009; Boehme et al. 2016), suggesting stream habitat degradation in my agricultural and urban study streams.

Temperature in forested sites was lower than in agricultural and urban sites throughout the sample period. This is likely due to the large riparian trees shading the stream in forested sites, as the riparian vegetation has been allowed to remain despite the presence of cattle grazing (Figure 17). The PFCs results reflect the importance of riparian vegetation in forested sites, which all scored at proper functioning condition compared to the other land uses, which had various categories of degraded condition. Agricultural sites had riparian vegetation dominated by reed canary grass and few trees so stream shading was minimal, consistent with other studies that found agricultural land use causes a distinct lack of riparian vegetation (Klemas 2014), and return flow from agricultural land use has also been found to increase stream temperatures (Younus et al. 2000). Urban land use followed the same temperature trends as agricultural land use, consistent with other studies (Pluhowski 1970; LeBlanc et al. 1997). A contributing factor to warmer urban streams could be urban infrastructure that can increase water temperature by as much as 5-8°C in summer (Pluhowski 1970). Finally, another problem with a lack of riparian vegetation is a lack of woody debris recruitment (Beschta and Taylor 1988; Allan 2004) that further contributes to reduced PFC.

In stream systems that support native salmonids which become stressed at 19° C (Gale et al. 2014; Jeffries et al. 2014), warmer temperatures are problematic and consistent with degraded habitat. Generally speaking, higher water temperature in streams leads to faster growth yet smaller size of macroinvertebrates, reducing food quality for salmon and other fish species (McCullough 2009). Also, salmon depend on temperature to know when to emerge as fry with higher temperatures causing early emergence (McCullough 2009), and high stream temperatures can also cause salmon to migrate upstream too early (Quinn and Adams 1996). Salmon and other fish species in these streams need specific temperatures and will not survive to mate if temperatures remain too high (Dittmer 2013). These disruptions to their lifecycle include the possibility of reduced reproduction and higher death (McCullough 2009).

A

B

Figure 17 Visual evidence of degradation. (A) Looking upstream at Cooke stream forest site, within a public forest where cattle have grazed the lower riparian vegetation. (B) Looking upstream of Naneum agricultural site.

All of these indicators are consistent with reduction in habitat quality in agricultural and urban sites in the streams I studied.

Urban land use had much lower concentrations of dissolved oxygen compared to forested and agricultural sites. Human impact and land use has been previously shown to reduce dissolved oxygen in Columbia Plateau streams (WA State Department of Ecology 2015). Low concentrations of dissolved oxygen in urban sites could have been influenced by warmer temperatures or underground stream reaches where photosynthesis cannot occur, specifically in Wilson creek which runs underneath Ellensburg, WA for a while then reemerges (Beaulieu et al. 2014). Additionally, biochemical oxygen demand could be higher in urban streams due to leaking sewer lines. Forested sites might be higher in dissolved oxygen for various reasons including lower water temperatures and more turbulent streams that introduce and hold more dissolved oxygen. In contrast, percentage saturation of dissolved oxygen did not vary by land use. This result is not uncommon as other research has found that measuring dissolved oxygen can be difficult due to factors such as weather influencing the results (Moerke and Lamberti 2004; Sliva and Williams 2016). Even though careful effort was taken to sample from 11 AM to 3 PM to avoid differences in concentrations because of daylight and photosynthetic activity, differences between days could not be accounted for.

Stream pH did not vary by land use, and all the measurements were between 6.5 and 8.5. In fact, the individual streams had more of an effect on pH than land use, possibly due to geologic factors specific to each watershed (Bailey et al. 1987). The lack of a land use effect is inconsistent with the observation that anthropogenic land use generally increases pH (Dow and Zampella 2000) and the phenomenon of increased pH in urban streams due to carbonate compounds released from concrete in urban infrastructure (Pluhowski 1970; Conway 2007). However, the lack of significant rainfall during my sampling might reduce runoff effects that can alter pH, as storm events have been shown to significantly increase pH via flushing of solutes from urban and agricultural areas (Zampella 1994; Dow and Zampella 2000).

Ammonium was not associated with land use differences in this study. This is inconsistent with other studies that found that urban (Zampella 1994; Berger et al. 2017) and agricultural streams (Quinn 2000; Johnson et al. 2003; Sliva and Williams 2016) with higher ammonium concentrations. It is possible that the forested sites, which in other studies have low ammonium concentration (Berger et al. 2017), have somewhat higher ammonium concentration due to agricultural land use including cattle grazing in riparian areas and logging, activities which can increase ammonium (Sliva and Williams 2016). However, streams in the western United States on volcanic bedrock such as those that I studied frequently have low N concentrations and a prevalence of nitrogen fixation associated with higher phosphorus concentrations (Johnson et al. 2003). On the other hand, there may have been technical issues that compromised my ammonium samples. Many old bottles broke while in the freezer before testing, and they leaked when thawing, potentially contaminating some of the samples.

Nitrate and phosphate varied by land use with forested sites having lower concentrations than urban and agricultural sites. Agricultural areas receive fertilizer to efficiently grow crops and organic waste from cattle farming (Chang et al. 2013) and this causes nitrate and phosphate to readily run off the landscape through return flow from irrigation (Wilcock 1986; Smith et al. 1993; Wernick et al. 2007). Furthermore, agricultural land use within forested sites had predominantly forested riparian buffers that decreased the amount of nitrate and phosphate entering the stream compared to grass riparian buffers more commonly found in agricultural and urban sites (Osborne and Kovacic 1993). Riparian buffers are effective because they slow runoff and increase the time for riparian plants to take up nitrate and phosphate before it enters the waterway. As seen in many other studies (Regetz 2003; Lehrter 2006; Shupe 2017), agricultural and urban land use is associated with elevated nitrate and phosphate concentrations in the Kittitas Valley, and these are indicators of reduced habitat quality.

EPT and HBI were significantly affected by land use in this study with forested sites having high EPT percentages and HBI ratings of “Excellent” meaning that pollution is not likely (Hilsenhoff 1988) and habitat quality is better. Substrate diversity in forested sites, woody debris, and large amounts of bank stabilizing riparian vegetation increase habitat heterogeneity which generally leads to higher habitat quality (Negishi et al. 2002). Agricultural and urban sites showed indicators of comparatively lower habitat quality with much lower EPT percentages and HBI ratings of “Good” to “Fairly Poor” respectively, associated with the likelihood of substantial pollution or probable organic pollution respectively (Hilsenhoff 1988). The EPT percentage in agricultural and urban areas dropped significantly partly due to the complete absence of order Plecoptera in two agricultural sites in all sample times. One exception was the Reecer urban site that was located at the beginning of a large restoration project where substrate was added to create habitat for more pollution intolerant species. While it is possible this improved habitat for stream insects, others have found that substrate additions will not necessarily increase habitat quality unless the structure and function of the stream ecosystem is restored (Moerke and Lamberti 2004). Agricultural and urban sites also contained large amounts of invertebrates including: *Tricladida Dugesiidae* (planaria), *Opisthopora* (terrestrial worms)*, Amphipoda Gammaridae* (scuds), *Isopoda Asellidae* (aquatic sow bugs), all of which indicate pollution (Cortelezzi et al. 2018). These findings reflect many other studies that find urban and agricultural land uses associated with reduced pollution intolerant macroinvertebrates (Negishi et al. 2002; Arnaiz et al. 2011; Berger et al. 2017; Burdon et al. 2017). The condition of the insect communities in my study sites supports the other findings of habitat deterioration associated with urban and agricultural land use in the Kittitas Valley.

Another potential explanation for the patterns I saw in my data is the river continuum concept. This concept states that biological processes change in the stream as it moves along a longitudinal downstream gradient (Vannote et al. 1980). All variables in this study could have been affected by this process because all streams began in forested areas and flowed through agricultural and then urban areas. This concept explains that headwater streams are typically smaller and faster flowing with larger substrate and lower temperatures. While this explanation could explain some of the variation in the data, I do not believe that this explains all of the variation in the data because this concept includes all stream orders (Vannote et al. 1980), while my study focuses on first and second order streams.

Temporal and spatial differences within the sampling strategy I chose could have caused variations in the data. Irrigation canals that cross all of these streams allow for the exchange of water. This could account for the fact that land use did not have an effect on some response variables includes pH, turbidity, and percentage dissolved oxygen. Rivers in this area are affected by the annual early September “flip-flop” where water released from certain reservoirs is decreased and water released from other reservoirs is increased (Bureau of Reclamation 2020). The purpose is to stabilize water flow in the upper Yakima River for spring Chinook salmon spawning. This “flip-flop” did not directly affect the streams in my study, but it might have affected the accumulation of water quality in the Yakima river from the streams in this study. For example, I predicted an accumulation of temperature in the Yakima River, but the increased reservoir water may decrease the temperature and negate the effects of these streams on the Yakima River. Additionally, a bout of heavy summer rain during the second week of August could have caused increased runoff into the stream and affected some of the water quality parameters I measured. Also, the nature of hay farming means that during hay cutting water delivery is decreased and there is less return flow. This means that if irrigation was not happening by chance during a sampling day, I would not have captured the true effect of irrigation, which was the reason I sampled in summer.

Conclusions

The effect of agricultural and urban land use on water quality of Kittitas Valley streams is apparent. I measured general degradation in habitat condition as streams move from forested to agricultural and/or urban land use. Channelization of the streams in agricultural and urban areas has caused downcutting, erosion, and simplification of the substrate composition. Riparian areas in these land uses have invasive reed canary grass that does little to shade the streams, thus increasing stream temperatures and reducing recruitment of LWD important for stream habitat heterogeneity. Higher nitrate and phosphate concentrations associated with agricultural and urban activity can degrade water quality in my study sites and possibly further downstream. Collectively, these factors have decreased heterogeneity of aquatic macroinvertebrate habitat, evidenced by a lack of intolerant species like the absence of the order Plecoptera in a few of the agricultural sites for all three sample times. Land use is known as a major factor that contributes to water quality and stream habitat degradation. My study highlights how land use in the Kittitas Valley is associated with degraded stream habitat with biological consequences observed in the aquatic insect community and the possibility of consequences for fisheries.

Management Recommendations

Management of the agricultural and urban land within this watershed would be most effective with best management practices (BMPs) that could mitigate or even reverse negative land use effects (Gabel et al. 2012). In agricultural areas, fencing out livestock, using no till farming, and restoring adequate riparian buffers are all BMPs (Gabel et al. 2012) that could be implemented at many of these sites. Native riparian vegetation that can help control temperature, stabilize banks, and add woody debris to the stream would help to restore ecological function and structure to the stream (Moerke and Lamberti 2004). Naneum agricultural site with riparian vegetation dominated by reed canary grass is reaching temperatures of 21°C in August and throughout the study period have a distinct lack of the order plecoptera. The best way to manage temperature is to plant native riparian buffers to shade out the stream. Short term this will not restore the conditions needed to support the order plecoptera, but as stream temperatures decrease and the channel is changed by stabilizing vegetation there could be a return of this order of macroinvertebrates (Peipoch et al. 2015). Many of these streams run through fields that are being farmed so implementing no till farming or fencing out cattle will reduce excess nitrate and phosphate from entering the stream.

In urban areas BMPs include wet and dry retention ponds, swales, and infiltration systems (Muthukrishnan et al. 2006) that retain stormwater to be to reduce delivery of nitrates and phosphates and if planted with native vegetation can reduce the temperature of the stream. Agricultural and urban sites both had stream reaches that ran alongside a road. For example, Cooke agriculture just upstream of the sample site was confined by a road on one side and cattle farming on the other where the cattle had full access to the stream. Fencing out the cattle would greatly reduce nitrates and phosphates from entering the stream and building a swale planted with native vegetation would decrease specific conductivity by trapping runoff from the road. This would also decrease the temperature of the stream by shading it out. Wilson and Reecer urban sites have large trees that shade out the stream, but the riparian buffer is not wide enough to keep runoff from increasing specific conductivity. Cooke urban site on the other hand has a relatively wide riparian buffer but is highly channelized increasing the transport of excess nitrates and phosphates from the agricultural site upstream. If these methods are implemented on the reaches of streams in this study, water and habitat quality could be improved, which would allow healthier stream habitat with the potential to improve survival of anadromous and resident fish species that are such important natural, economic, and cultural resources (Regetz 2003).

To adequately manage this land to improve water quality, every stakeholder must be included in the management plan. The documented effect of the accumulation of pollution from small order streams draining into larger order streams (Alexander et al. 2007; McDowell et al. 2017) means that managing these streams involves a large number of stakeholders in a large area. For example, pollution from streams in this study could accumulate in the Yakima river within Yakama tribal lands (Alexander et al.

2007). Therefore, this management plan would have to include the Yakama Nation; the farmers who own the land these streams run through; people living in Kittitas County; government officials, including Department of Fish and Wildlife; Department of Natural Resources, Ellensburg and Kittitas city officials; and Kittitas Reclamation District and other irrigation companies that manage the irrigation infrastructure that interconnects with these streams.

Cultural and social constraints require the management of water resources within the context of the watershed as the only way to achieve realistic goals (Moerke and Lamberti 2004). Streams will likely not return to their historical state, but BMPs and an inclusive management strategy could greatly improve water quality for a diverse set of ecosystem services that benefit social causes. I recommend bringing together the various water management stakeholders in this watershed to create a comprehensive and realistic plan of using BMPs to improve water quality in agricultural and urban land uses observed in this study.

Improvements

There were some improvements that could have been made to this study. The first is that not all the streams flowed through urban land uses, so the study design was not fully crossed, making statistical analysis a challenge. Moreover, the relatively small sample size might have influenced the statistical analysis through outliers or not capturing a strong enough signal. I decided on a 100 m buffer to evaluate land use along these streams because many of them are small and there is evidence from previous studies that smaller order streams are influenced more by local land use than upstream watershed land use (Buck et al. 2004; Feld 2013). With that said there is also some evidence that watershed level land use analysis gives a better indication of the effects of land use on water quality (Sliva and Williams 2016). Nevertheless, the consistency between many of my findings and the literature on land use influences on water quality suggest that my use of a 100 m buffer to quantify land use was justified. Another problem that occurred in this study was that some of the ammonium sample bottles broke in the freezer before I could perform the analysis, potentially contaminating my samples. Using more durable plastic containers would have mitigated that problem. Because I could not measure all 14 sites in one day with the restriction of sampling between 11 AM and 3 PM, changes in the weather from day to day could have caused uncontrolled variation in the data collected. Finally, these samples were collected during the summer, so I do not have a complete picture of how land use affects streams on an annual basis. Land use effects from rain runoff or snowmelt could reveal more or different relationships compared to what I found during summer.

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