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Spatially explicit ecosystem service valuation (ESV) allows for the identification of the location and magnitude of services provided by natural ecosystems to human activities along with a measure of their significance based upon economic valuation. While ESV has been used to provide new insight into land use management, few studies have identified the connections between the values of ecosystem services and ecological sensitivity to nitrogen loading despite a growing body of ecosystem service literature. This research combines a GIS-based, value transfer approach to map ecosystem services in the Lower Yakima River Basin (LYRB), Washington, USA, along with estimates of nitrogen loading to identify how nitrogen management may affect ecosystem services in the basin. This analysis combines values of ecosystem services with estimates of nitrogen loading and identifies subwatersheds and specific parcels within a Groundwater Management Area (GWMA) most susceptible to reductions in ecosystem services due to excess nitrogen loading. Based on the benefit transfer analysis, wetlands and forested areas have disproportionately high values of ecosystem services when compared to their land area in the LYRB, while pasture and cultivated crops contribute much less to the total value of ecosystem service flows in proportion to the total area in the LYRB. Across the study area estimated nitrogen loads are strongly driven by the

location of concentrated animal feeding operations (CAFOs) and cultivated crops. Areas of particularly high nitrogen loading and high ESV may highlight specific areas for achieving immediate success in increasing or maintaining ecosystem services through appropriately focused regulatory mechanisms. The land cover analysis however, completely neglects the values and importance of subsurface processes and groundwater resources in ecosystem service assessment, and therefore an econometric model is applied to estimate willingness to pay (WTP) to maintain safe nitrate levels in private wells. Through the incorporation of WTP estimates for groundwater quality, a more complete economic and ecological perspective on the effects of landscape N loading in the study site is highlighted. The results of these estimates clearly indicate that ecosystem services from groundwater should be considered to have significant value in the LYRB.

Further economic valuation data on specific land cover types and the value of groundwater quality, whether from primary studies or meta-analysis, is needed to refine relative measures of ecosystem service values and more confidently describe these values in specific dollar amounts. Additionally, limits in spatial data resolution may contribute to errors in location and magnitude of ecosystem services, and is an area in need of further development. Despite these potential limitations, this analysis highlights a promising direction for combining spatially explicit ecosystem service valuation with nutrient loading data to identify the location and potential magnitude of effects on ecosystem services from management practices.

An Ecosystem Service Approach to Inform Reactive Nitrogen Management in the Lower Yakima River Basin, Washington.

by

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Morgan Crowell, Author

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Section 1. Introduction

The relationships between land use, ecosystem productivity and human health have only become more apparent as the world faces a future full of climate uncertainty, population growth and enormous technological capabilities. Particularly with regard to agricultural productivity and water quality, biogeochemical processes are critical to a sustainable and healthy future for all living things, and efforts to ensure human health and a safe environment are well intentioned to investigate how human activities continue alter these natural cycles with particular attention towards maximizing human and environmental benefits.

To date, a significant emphasis of academic literature and regulatory efforts have focused on single objective approaches using a single metric to measure environmental quality or human health risk. While this approach has served a valuable function in terms of protecting environmental quality and human health and driving development of beneficial management practices and technologies, as the interconnectedness of Earth systems is increasingly recognized within the regulatory and academic communities, the development of multiple metrics for assessing land use, environmental quality and benefits from natural ecosystems provides a promising direction to inform land use decision-making processes. Understanding the simultaneous delivery of multiple benefits in a variety of forms from ecosystems may allow for optimized land management practices that protect human and environmental health, rather than focusing on a single goal at the potential expense of additional benefit opportunities.

This research attempts to address the need for the development for multiple metric approaches to understanding land management by applying an ecosystem service approach to reactive nitrogen in the Lower Yakima River Basin (LYRB) in Central Washington. The ecosystem service approach attempts to incorporate multiple benefits delivered from ecosystems into measures of value that can be used in comparative analysis to current land use practices. In terms of academic contributions, this research sought to determine the following:

- How can an ecosystem service approach be applied to the issue of reactive nitrogen, and what potential outputs and conclusions can this approach be expected to produce?
- Based on the results of this analysis, what is the potential for this approach to be applied to other environmental issues in a diversity of study sites and scales?

- How does an ecosystem service approach adequately address the complexity of issues created through human alteration of biogeochemical processes? What may be areas where this approach fails, if any?

It was hypothesized that a statistical relationship between the location and magnitude of benefits from ecosystem processes and nitrogen inputs from human activities would exist. The hypothesized existence of high nitrogen loads in areas proximate to valuable ecosystems would indicate specific areas of concern and would serve as increased evidence for immediate actions to address potential environmental and health issues. While nitrogen moves with considerable complexity throughout the environment, demonstrating the spatial relationship between landscape nitrogen loading and valuable ecosystems would provide relevant, valuable information for stakeholders in the LYRB and may demonstrate a useful methodology for future application in other study sites.

In addition to theoretical, academic questions, this research was firmly rooted in practical application and important goals to the specific study site included:

- What is the potential for reductions in human benefits from ecosystem services from nitrogen loading in the LYRB? What are the locations of these areas and what ecosystems are present?
- What nitrogen sources and areas should be identified for priority efforts to address concerns over environmental alteration and human health risk due to nitrogen inputs?
- Do the results of an ecosystem service approach support existing regulatory efforts?

While academic research may rightly tend to focus on the development of rigorous research to advance scientific thinking and problem solving, this research is driven by a strong belief that immediate practical application is an equally important component of academic research endeavors, and this is reflected throughout. The balance between application and scientific advancement is a continuing process, and it hoped this research attains an appropriate balance of utility to both academic thinkers with an interest in ecosystem services and environmental regulators interested in alternative frameworks for improving human and environmental health through informed decision-making processes.

Background Section 2.1: Reactive Nitrogen and Agriculture

Nitrogen is a fundamental component of metabolic processes in every living cell. The productivity of ecosystems around the planet are tied to the availability of nitrogen and therefore the movement and transformation of nitrogen compounds have a critical effect upon the fundamental natural systems that support all types of life on Earth. Given the importance of nitrogen it is appropriate that it is present in large quantities in Earth systems. But despite its abundance, nitrogen is often found in a form unavailable for use by living cells. Nitrogen exists in atmospheric, terrestrial and aquatic systems in both reactive and nonreactive forms, of which only reactive nitrogen (Nr) forms are accessible for use by living cells. Nonreactive nitrogen (N_2) contains a strong triple bond that requires high-energy inputs or specialized microbes to break and convert N_2 into Nr as either inorganic reduced forms (NH_3, NH_4+) or inorganic oxidized forms (NO_X, HNO_3, N_2O, NO_3, NO_2). Once reactive nitrogen is formed, it is subject to a diverse variety of transformations and transport processes that allow a single atom of nitrogen to move through many different compounds and have multiple effects on natural systems and human health before it is transformed back into a nonreactive form through denitrification. The processes of denitrification in Earth systems along with the transport and fate of nitrogen have been the subject of a large body of research (Jackson et al. 2008, Korom 1992, Puckett et al. 2006, Seitzinger et al. 2006, Soares 2000) which has been critical in understanding how and when nitrogen transformation processes may occur. The policy implications, potential solutions and metrics for assessing the impacts of nitrogen loading is also represented in existing literature (Galloway et al. 2008, Mosier et al. 2001, Birch et al. 2011). The nitrogen cascade framework (Galloway and Cowling 2002, Galloway et al. 2003, Galloway et al. 2004) has further described the potential for multiple impacts to Earth systems and human health from a single nitrogen atom as it moves through the environment.

Historically Nr has been created through the natural processes of biological nitrogen fixation (BNF) in specific microbes and through the high energy of lightning. Before the alteration of the nitrogen cycle by human activities, the rates of nitrogen fixation and denitrification processes were approximately equal in environmental systems (Galloway 2003). However, nitrogen fixation processes have increased dramatically in the past century. The increase has been due to widespread cultivation of crops such as legumes and rice that increase BNF, the combustion of fossil and biofuels, and the introduction of the Haber-Bosch process of nitrogen fixation in the early 20th century with subsequent increase in fertilizer application rates. This has led to a situation where rates of nitrogen fixation exceed rates of denitrification, resulting in an increasing supply of Nr in Earth systems (Galloway and Cowling 2002). Reductions in denitrification processes due to land use changes, such as reduction in wetland areas, are also contributing factors to an increased supply of Nr (Galloway et al. 2003).

While the increase in Nr within Earth systems has a variety of causes, evidence of this increase has been well demonstrated (Galloway and Cowling 2002, Vitousek et al. 1997, Schlesinger et al. 2006). The impacts from the increase of Nr on human activities and ecosystem processes have been both positive and negative, but the dynamic of costs and benefits from increased Nr in the environment are poorly understood and researchers have hypothesized a threshold where benefit loss due to the effects of excess Nr may exceed benefits received (Galloway et al. 2003). Because living cells require Nr for metabolic processes, it is common in a variety of global environments that nitrogen availability is a limiting factor in biological growth. Intentional increases in Nr through fertilizer applications and BNF have largely sought to increase nitrogen availability and therefore increase growth and productivity in crops. The benefits from the increase of Nr in terms of increased crop production and food availability have been significant, and increased Nr from human processes has been a major factor in the increase in agricultural productivity in the 20th century. When considered with respect to an increasing global population, and potential deceases in agricultural output due to climate change (Rosenzweig and Parry 1994), the importance of Nr to increased production is clearly demonstrated. Additionally, while energy production and transportation also have contributed significantly to Nr increases (Galloway and Cowling 2002), the development of the modern global economy has a strong link to energy and transportation powered by fossil fuels and therefore increased environmental Nr should also be seen as a product of this economic development.

The increase in Nr production that has fueled economic development and agricultural productivity increases has not come without negative effects to both humans and natural ecosystems. In terms of environmental effects, increased Nr has contributed significantly to problems in atmospheric, terrestrial and aquatic systems. Examples of these effects include:

Atmospheric:

- increased greenhouse gas production contributing to global climate change
- stratospheric ozone depletion

Terrestrial:

- decreases in biodiversity and productivity if critical threshold of Nr is exceeded
- soil acidification

Aquatic:

- acidification of lakes and streams
- eutrophication
- hypoxia
- loss of biodiversity
- habitat alteration

These environmental problems often lead to or are accompanied by concerns over public health and human well-being. Public health concerns with increased Nr include:

- degraded air quality due to increased aerosols, ozone, particulates and allergenic pollen
- drinking water quality (particularly with regard to nitrate levels and acidification)
- algal bloom released neurotoxins

Along with health concerns there are also numerous other secondary effects that may not directly affect human health, but may impact quality of living in terms of recreation, economy and cultural values. For example, drinking water quality in a private well may be a health concern but it may also have negative impacts upon property values that could have larger scale societal implications.

Because of the enormous benefits of Nr to agricultural production, regions of intensive agricultural production are particularly significant in research related to Nr management. In many agricultural regions, anthropogenic Nr inputs come from relatively universal Nr sources such as atmospheric deposition, energy production, transportation, septic systems, household fertilizer use, and intensive non-agricultural land uses such as golf courses. But agricultural areas also typically experience large inputs in the form of Nr rich fertilizers and the process of Nr concentration that comes from animal wastes associated with ranching and concentrated animal feeding operations (CAFOs). Because of the increased Nr inputs, these areas provide an opportunity to examine impacts of Nr on environmental systems and human health in a more intense situation of nitrogen loading. Agricultural areas have also been found to suffer from numerous environmental and human health risks relating to high levels of Nr (Balazs et al. 2011, Harter and Lund 2012) but it is difficult to identify exact sources of a given nitrogen atom in a particular location because of the fate and transport processes in the movement of Nr. Therefore while there is a high demand to develop precise techniques for tracking nitrogen movement, particularly in subsurface processes, there is also a need to understand, on a regional scale, how estimates of nitrogen loading relate to regional ecosystems and how Nr management decision making processes may be informed by these relationships. Because of the more intense level of Nr inputs and a greater tendency for environmental concerns, it is also common for agricultural areas to experience conflict among stakeholders over affected resources. These areas offer excellent opportunities to examine the relationships between the spatial distribution of Nr inputs, regional ecology, stakeholders and regulatory frameworks in areas with a high diversity of complex nitrogen processes. These regions also offer a chance to examine the capacity for specific issues to serve as regulatory levers for Nr management that will be most efficient in addressing priority issues.

Background Section 2.2: Ecosystem Services

The impacts of Nr in the environment and in terms of human activities are broad and require a multidisciplinary approach to understand how Nr management may create benefits and costs on a regional and global scale. Understanding how Nr management alters ecological functions and how these changes may affect human populations is a critical first step in identifying multiple metrics and key areas for stakeholder cooperation that can be applied in Nr management decision-making. Particularly given the uncertainties and complexities inherent in Nr processes, general frameworks and multiple methods of valuation may provide a valuable approach that does not require a high level of precision or quantification to demonstrate key findings that can aid in efficient, integrative policy and management solutions. The concept of ecosystem services provides a useful framework that can be applied to Nr management to evaluate how Nr inputs may impact human activities, and potentially aid in the identification of benefit thresholds for Nr levels.

The concept of ecosystem services is rooted in the fundamental role of natural systems and processes to support human activity on a global scale. From air and water quality to climatic regulation of our atmosphere, the services provided by these geophysical, ecological and atmospheric processes are colloquially referred to as ecosystem services, with a nearly standardized definition as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Daily 1997). The concept of ecosystem services has been the subject of extensive studies and applications in the past 15 years, but even the precise definition and typologies for ecosystem services is a continued area of analysis (Haines-Young et al. 2009, Boyd and Banzaf 2007, de Groot et al. 2002). The classification of ecosystem services created by the Millennium Ecosystem Assessment Board (MA 2005) defines services as “supporting, provisioning, regulating or cultural” to assist in conceptualizing, defining and valuing these services. While this classification system is widely recognized as a valuable tool in the establishment and integration of ecosystem services into resource management and environmental economics, there have also been efforts to develop classification systems that focus on defining ecosystem services in a specific decision making framework in terms of intermediate and final services (Fisher et al. 2009).

With particular relevance to agricultural areas, research (Swinton et al. 2007, Antle and Stoorvogel 2006, Power 2010) has attempted to understand the relationships between agro-ecosystems and ecosystem services, but has not established a uniform framework for doing so. Previous research has also suggested the concept of 'ecosystem dis-services' to represent the role of excess nutrient inputs, habitat loss, and introduction of pesticides among other factors (Zhang et al. 2007). Despite the difficulties to specifically define and classify what ecosystem services are and to incorporate agriculture in an ecosystem service framework, there has been sufficient research conducted using the concept of ecosystem services that an 'ecosystem services approach' can be synthesized and applied (Compton et al. 2011, Mark 2011, Zhang et al. 2007).

Approaching an issue such as nitrogen management from a ecosystem service perspective may more fully account for a wide range of benefits and interconnected relationships than a traditional single metric approach focused solely on a particular desired environmental or public health outcome. By focusing on multiple components of ecosystem benefits, it may be possible to create management plans, policies and regulations that incorporate a holistic perspective of multiple simultaneous ecologically connected benefits. However, the methods for integrating the ecosystem services framework into decision-making process are unclear, and largely more a subject of theoretical debate as opposed to manifested management and application case studies. Critical assessments of the ecosystem framework (Simpson 2011), and reviews of various ecosystem services typologies and categorizations (Haines-Young et al. 2009) highlight the lack of cohesion in even the most well cited ecosystem service literature and make it clear that the actual application of an ecosystem services framework in decision-making is still in significant need of development.

To date one of the most common approaches to the integration of an ecosystem service framework has been to use economic methodologies to estimate ecosystem service benefits to humans for comparative analysis in decision-making. Costanza et al. (1997) presented one of the earliest uses of economic valuation of ecosystem services, and while the methods and precision of this work have been the subject of a great deal of debate, it helped to establish a precedent that has impacted ecosystem service research and economic valuation. The methods and models for ecosystem service valuation have subsequently evolved dramatically in the past 15 years and represent a considerable body of work (Liu et al. 2010, Martin and Blossey 2009, Sagoff 2010, Turner et al. 2010, Winkler 2005).

The most fundamental step in the valuation of ecosystem services is the definition of the service/good/benefit to be valued and the beneficiary. While this may be an intuitive assumption many studies do not specify this required information for a valued service/good.

Any given ecosystem may provide a wide variety of benefits to humans and therefore identification of these benefits is a necessary component to valuation. When a specific service/good has been identified it is typically determined to be either a market or non-market service/good that can be valued using one of the examples of appropriate methods below:

Market Good/Service

- Pricing Observation

Non-Market Good/Service

- Revealed Preference Methods
    - Hedonic Pricing
    - Travel Cost
    - Damage, Replacement or Substitute Costs
- Stated Preference Methods
    - Contingent Valuation
- Benefit Transfer (including meta-analysis)

While the performance and precision of these valuation methodologies for ecosystem service is a subject of research on it's own right (Plummer 2009, Brouwer and Spaninks 1999) this section simply introduces the variety of valuation methodologies and the market/non-market service/good differences to reinforce the complexity of valuing a variety of diverse benefits that may be delivered by a single ecosystem. This is further complicated when human capital investments are considered with regard to benefits from ecosystems. Because capital investments must be accounted for as part of ecosystem valuation, it is necessary to differentiate between benefits from ecosystems which have been engineered for specific benefits through human capital investment, and primary ecosystems which deliver benefits to humans without receiving any capital investments to generate these benefits. We can separate these by defining "ecosystem services" as benefits to humans from primary ecosystems receiving no human capital investments, and "engineered ecosystem benefits" that require capital investment to create benefits. We can also integrate the core concept of intermediate/final services (Fisher et al. 2009) and ecosystem services and dis-services (Zhang et al. 2007) into this conceptual model as shown in Figure 1. This model is helpful in that it also accounts for feedback relationships between engineered ecosystems and primary ecosystems. A primary goal of an ecosystem service approach to land management seeks not only to understand the properties of primary ecosystem services, but also to maximize positive feedback between primary and engineered ecosystems to maximize human benefit. Examples of ecosystem services/dis-services and engineered ecosystem benefits are shown in Table 1. Previous work to incorporate accounting for human capital investments in ecosystem services (van Houtven and Sinha 2012) has proposed alternative frameworks to those shown in Figure 1, however these alternatives are seen as components in more complex economic metrics and therefore do not offer a simple, accessible, conceptual framework for discussion, development and application. A final component of this model in need of mention is that human actions may have negative impacts on both primary and engineered ecosystems that must be accounted for in assessment of ecosystem services and engineered ecosystem benefits.

Combining an ecosystem services approach to nitrogen management allows for the integration of the complex fate and transport processes of nitrogen throughout multiple ecosystems that is critical to nitrogen management. The nitrogen cascade illustration in Figure 2 (Compton et al. 2011) is a useful conceptual model for understanding how nitrogen moves through various Earth systems, and where sources of reactive nitrogen are added in these systems. It also illustrates potential pathways and how nitrogen compounds move throughout Earth systems to show the relationships between nitrogen sources, the nitrogen cascade framework and human activities. Because of the movement of nitrogen through atmospheric, aquatic and terrestrial systems, management and regulation through the nitrogen cascade is also separated into different regulatory authorities.

In the United States, the primary regulatory mechanisms relating to nitrogen management are the Clean Air Act of 1970, the Clean Water Act of 1972 and the Safe Drinking Water Act of 1974. Despite the fact that these regulations were enacted decades prior to the development of the ecosystem services concept, the core ideas of ecosystem services, (that natural systems provide valuable services and goods to support human activities) are implicit in each of these regulations. The Clean Air Act was established to “protect public health and public welfare” (EPA 2012). The Clean Water Act to ensure a “designated use” such as drinking water, recreation, etc., and the Safe Drinking Water Act was established to “protect public health by regulating the nation’s public drinking water supply” (EPA 2012). All of these regulations essentially function to protect particular regulating, provisioning and cultural ecosystem services across multiple ecosystem service classifications that benefit human activity in the United States.

While the connections among ecosystem services, nitrogen and regulatory decision-making is clear, there are still few tools for assessing ecosystem services with regards to specific pollutants or nutrients such as nitrogen. Spatially explicit methods of ecosystem service valuation continue to be developed and refined (Nelson and Daily 2010, Tallis and Polasky 2009, Sherrouse et al. 2011), but these do not focus on individual pollutants and ecosystem services for the purpose of informing management decision-making. By connecting methods of ecosystem service mapping with economic valuation and nitrogen loading estimates, it will be possible to show where ecosystem services are being provided, and estimate value for these services and how nitrogen loading levels may be impacting them. This ecosystem service approach is clearly compatible with the goals of existing regulation and will provide a valuable service to informing management decision-making. This research seeks to demonstrate a general methodology for the application of this approach in a large agricultural basin to estimate the general distribution and values of ecosystem services and evaluate the potential impacts from nitrogen management. An additional goal of this research is to demonstrate how such an application can be completed with limited resources as a first step for determining the appropriate direction, location and necessity for comprehensive valuation and policy assessment.

Figure 1: Ecosystem Service Framework Conceptual Model

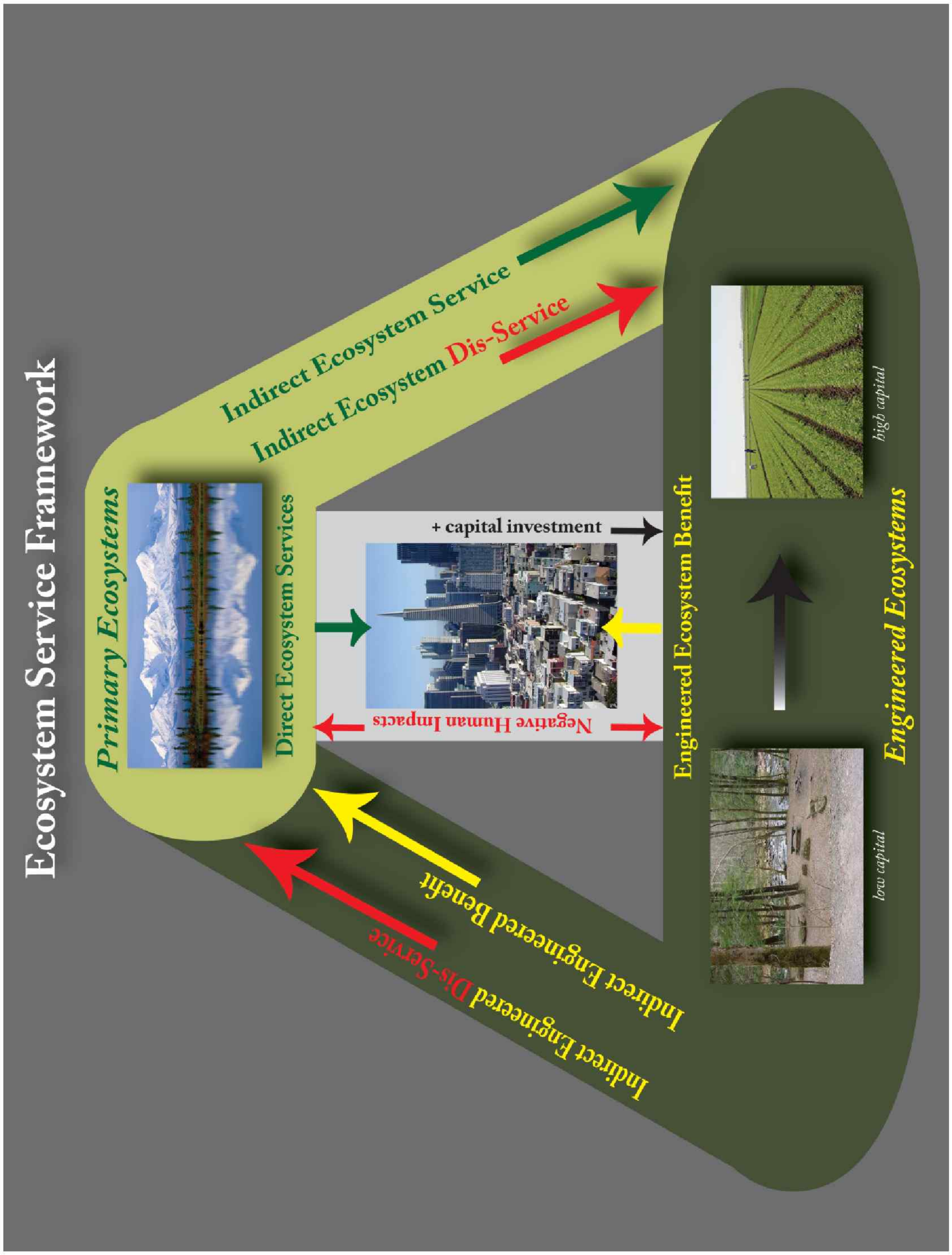
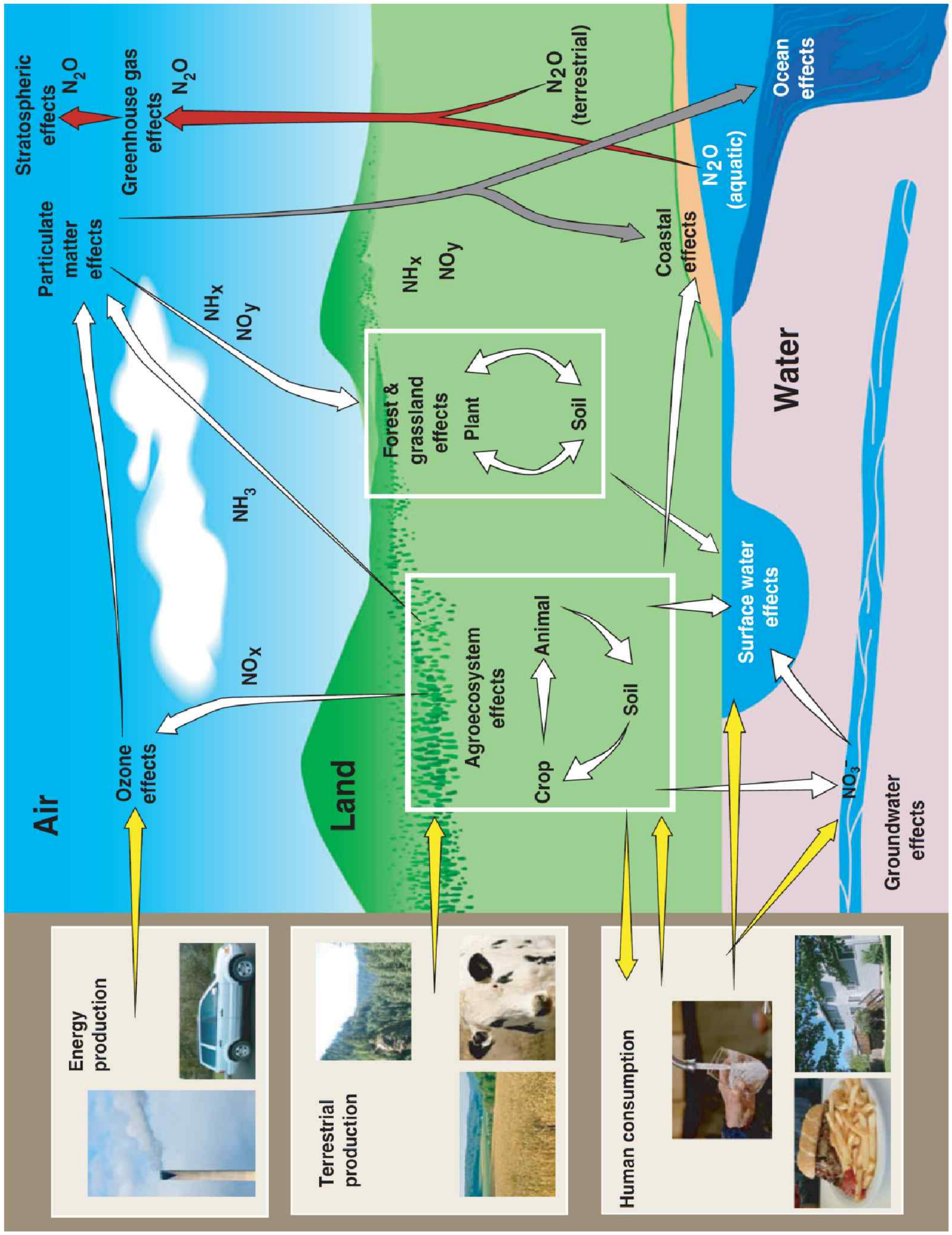


Table 1: Ecosystem Service Framework: Selected Examples

Ecosystem Service Framework: Selected Examples	
Primary Ecosystems *Direct Ecosystem Services*	***Engineered Ecosystems*** *Engineered Ecosystem Benefit*
Non-Market Goods/Services	
climate & atmospheric regulation beneficial plant/animal disturbance prevention recreation water supply water regulation cultural, educational & spiritual values	**climate regulation soil conservation habitat pollination nutrient retention water supply water regulation cultural, educational & spiritual values**
Market Goods/Services	
raw materials food medicinal genetic ornamental carbon sequestration* watershed services*	**raw materials food fuel recreation waste treatment**
Indirect Ecosystem Services	*Indirect Engineered Benefit*
climate regulation biogeochemical regulation pollination water supply soil processes genetic material	**climate regulation biogeochemical regulation pollination soil processes**
Indirect Ecosystem Dis-Service	*Indirect Engineered Dis-Service*
pest damage resource competition	**habitat loss resource competition excess chemicals/nutrients disruption of soil processes**

**markets under development*

Figure 2: Illustrated Nitrogen Cascade Compton et al. 2011



Background Section 2.3: Study Area

The study site for this research is the Lower Yakima River Basin (LYRB) or Washington State Department of Ecology Water Resource Inventory Area #37 in Central Washington State as shown in Figure 3. The Lower Yakima River Basin is a 749,908-hectare watershed that annually receives a mean of approximately 114 centimeters of rain at higher elevations and 15 centimeters at the valley floor near Kennewick (Ely et al. 2011). This semi-arid climate receives most precipitation during the winter months, typically in the form of snow at higher elevations. Along the northern edge of the LYRB in the valley floor, the city of Yakima (the largest population center in the LYRB) experiences an average summer high temperatures reaching 88°F with an average low temperature in the winter reaching 21°F (Yakima 2012).

The valley floor consists of quaternary flood deposits, loess and small areas of non-marine sedimentary rock. Areas immediately outside of the valley floor make up a majority of the LYRB and are part of the Colombia River Basalt group. Higher elevations along the western boundary of the LYRB also include areas of Quaternary and Pliocene volcanic rock (Vaccaro 2009). The lower basin formation and deposits are primarily associated with the draining of Lake Missoula an estimated 19,000 to 13,00 years ago (WA-DOE 2006, Newton 2010). Groundwater is found in confined, unconfined, semi-confined and perched conditions. Groundwater in unconfined shallow sedimentary aquifers in the valley floor tends to circulate laterally towards the Yakima River and it's tributaries unless intercepted by local wells. Seasonally, in certain areas, the groundwater levels may drop sufficiently to induce flow loss from surface water to the shallow aquifer (Ely et al. 2012). Most deep wells are drilled into confined aquifers in the Columbia River Basalts, however due to stratigraphic variations, this formation may be found at depths shallower than the shallow sedimentary aquifer. Because of the high level of agriculture and irrigation, recharge is estimated to range from 17.8 to 63.5 cm per year in irrigated areas to less than 2.5 cm in nonirrigated areas. Mean annual recharge from 1950-2003 is estimated at 39.6 cm including irrigation effects, and 30.2 cm if irrigation was not present (Vaccaro 2009).

The Yakima River enters the LYRB through the city of Yakima and Union Gap before flowing southeast to the confluence with the Columbia River just below the city of Richland. The estimated mean annual streamflow for the Yakima River, taking into account diversions and returns, is 101.9 m^3/s, while the estimated unregulated mean annual flow is

158.6 m^3/s (Vaccaro 2009). During the irrigation season of March to October, about 45% of the water diverted for irrigation is returned as streamflow and these returns account for about 75% of stream flow during the summer months in the lower basin near Parker (Vaccaro 2009).

The LYRB is primarily located in the Columbia Basin ecoregion, with a small area in the Eastern Cascade Slopes. Native vegetation in most of the area in the lower basin is big sagebrush and bluebunch wheatgrass, with Ponderosa pine and Douglas fir at higher elevations (WA-DOE 2006). Aquatic ecosystems, predominately riverine and wetland environments, support a wide variety of fowl and fish populations, both of which experienced significant declines post Euro-American settlement. Historic fish populations included spring, summer and fall Chinook, Coho, sockeye and steelhead, with current hatchery programs actively working to restore many of these salmon and steelhead runs. Currently the bull trout, summer steelhead and spring Chinook are listed under the Endangered Species Act, with native populations of Westslope cutthroat trout, redband rainbow trout and Pacific Lamprey listed as "under concern" by the U.S. Fish and Wildlife Service. Summer/fall Chinook, sockeye salmon and reintroduced Coho salmon are also found in the region (USFW 2012).

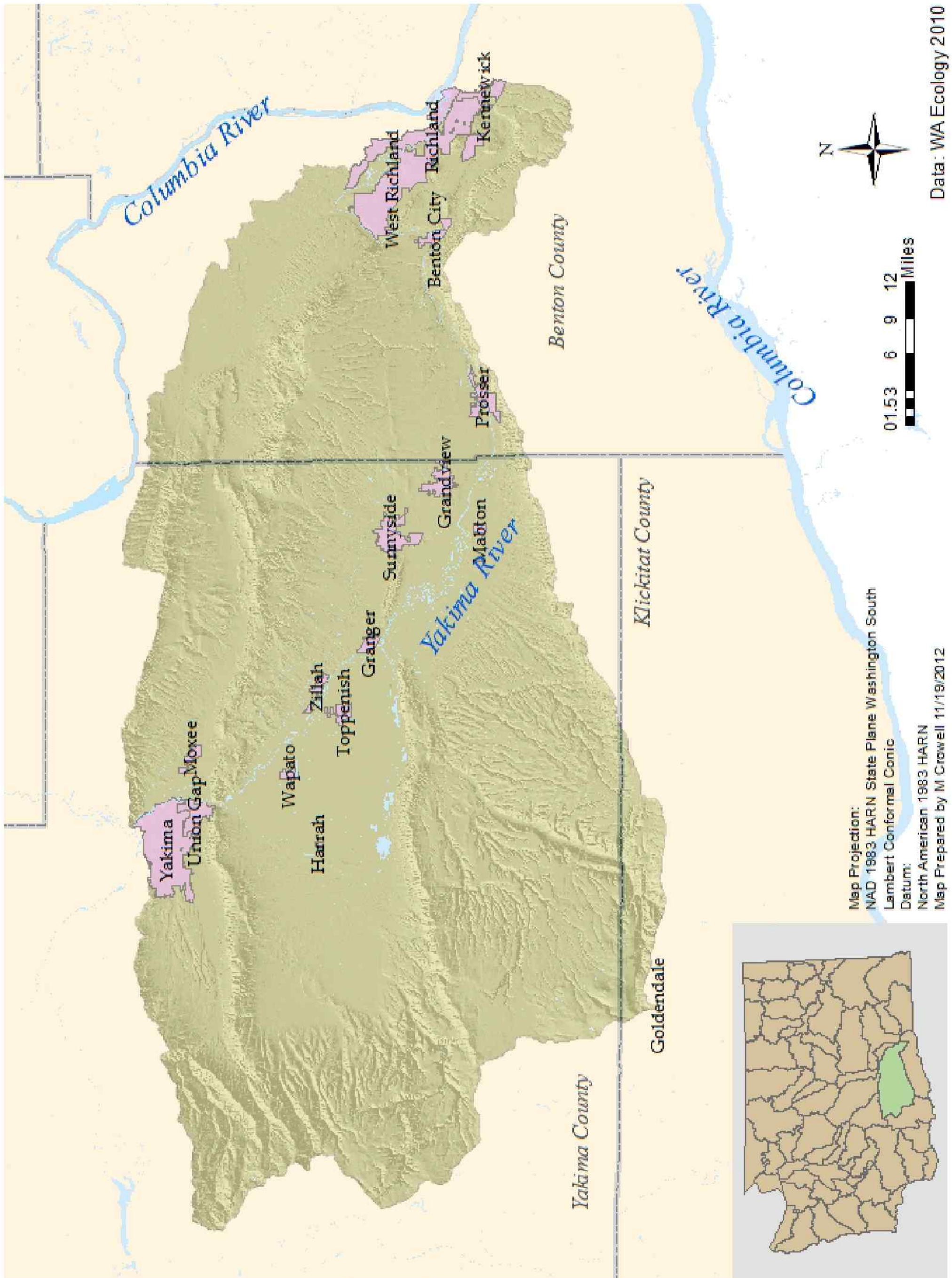
The LYRB supports a population of approximately 250,089 people and while the primary population centers are Yakima, Sunnyside and Toppenish, a significant portion of the population lives in unincorporated areas (WA-DOE 2010). Established in 1855, the Yakama Nation currently retains sovereignty over 48% of the land area in the LYRB and has been actively working to restore the quality and quantity of surface water and riparian habitats aimed at restoring salmon runs. Land ownership throughout the rest of the LYRB is 36% private, 12% federal 4% state and less than 1% local ownership. Seventy-four percent of the LYRB is in Yakima County, with Benton County (24%) and Klickitat County (2%) in the south and southeastern parts of the basin (WA-DOE 2006). In Yakima County, 21.8% of the population was below the poverty level (2006-2010) with a median annual household income of $42,877. The county population is 46.9% white, not of Hispanic origin, and 45.8% Hispanic or Latino. Yakima County is 5.6% Native American, which is over three times the percentage found in all of Washington State (1.8%) (Census 2010).

With 151,036 hectares in cultivated crops (Fry et al. 2006), the Yakima Valley produces over $1.2 billion annually in agricultural products and is the largest producer of hops, mint, apples and milk/dairy products in the US (YCDA 2012). While certain industries such as dairy and wine grapes have expanded in the region only more recently, the LYRB has a long history of agriculture and irrigation dating back to the late 1800s (Ely et al 2012). Beginning predominantly with irrigation diversions from the Yakima River, by the summer of 1905 these diversions for the first time caused the river itself to nearly dry up in the LYRB, with the Sunnyside irrigation canal being the largest dam/canal system located just south of the city of Yakima. As the number of dams and canal systems increased in the LYRB, native salmon fisheries have been decimated and all aspects of surface water quality have been affected. Working together, the efforts of state, tribal and federal agencies along with local irrigation districts have resulted in the establishment of Total Maximum Daily Loads (TMDLs) for DDT and turbidity in the Lower Yakima River, along with a TMDL for fecal coliform for the major irrigation outflow at Grainger Drain. Other TMDLs currently under development for drains, tributaries and the Yakima River in the LYRB include total phosphorous, temperature, fecal coliform and toxics (WA-DOE 2012).

Because of the visibility of the human impacts on surface waters and the importance of salmon to the significant tribal population in the LYRB, actions towards improvements have been for forthcoming and will likely continue for decades to come. Public attention has been much slower to notice and respond to issues related to groundwater resources, with the first major project towards assessing and understanding groundwater resources in the area not beginning until the 1990's (Ely et al. 2012). These initial efforts focused solely on groundwater quantities, and it was not until 2008 that significant public and institutional attention became focused on groundwater quality and specifically nitrate levels. Because groundwater from private wells is estimated to be the primary source of drinking water for approximately one-third of LYRB residents (WA-DOE 2010) the ecosystem services regulating water quality and the nonmarket good of safe drinking water are of significant interest to local, state and national stakeholders. Recent sampling of wells in the LYRB provides an estimate that 12% of wells exceed the nitrate-N MCL of 10mg/L (WA DOE 2010).

With increases in dairy production over the last 20 years and extensive use of fertilizers dating back at least several decades, agricultural activity is a clearly a contributor to increased reactive nitrogen in the LYRB. However, abandoned or improperly sealed wells, septic systems, golf courses, hobby farmers, private landscaping, non-commercial livestock and atmospheric deposition of nitrogen are also important nitrogen sources, and therefore determining the most efficient and equitable methods for managing nitrogen for groundwater quality is a very complex question. Given the decadal timescales potentially associated with nitrogen movement in groundwater, legacy nitrate sources such as fertilizer applications from previous crops like sugar beets, also present a challenge to both physical scientists and land managers. Because of the focus of nitrogen management efforts has been unclear, Washington Department of Ecology along with input from Yakima County and the US EPA have created a Groundwater Management Area (GWMA) which will be tasked with determining the best path of action for addressing ground water quality issues in the LYRB. Approved in fall 2011, the GWMA received official state funding in Spring 2012 and has initiated several meetings of the Groundwater Advisory Committee.

Figure 3: Study Area



Methods Section 3.1: Land Cover Analysis

A first step in understanding ecosystem services for a given area is an evaluation of the study site ecology, land cover types and relevant ecosystems. While ecosystem services are defined by their benefits to humans, understanding the natural environment within the study site is a critical step before evaluating how these ecosystems may provide services to relevant human populations. With the availability of land cover data based on satellite imagery, a GIS-based land cover analysis provides a simple, general method for evaluating the types of ecosystems present in the LYRB. The land cover analysis approach in ecosystem service mapping has been used in a wide variety of study sites and is well documented in the existing literature (Troy and Wilson 2006, Liu et al 2010, Costanza et al. 1997, Eade and Moran 1996, Kreuter et al. 2001, Lovett et al. 1997). In much of the research utilizing ecosystem service mapping from land cover data, significant issues are limitations in resolution and data quality. To address these limitations in this research, multiple datasets have been evaluated and combined into a single aggregated dataset that ranks various datasets based upon the relevance to this research and precision.

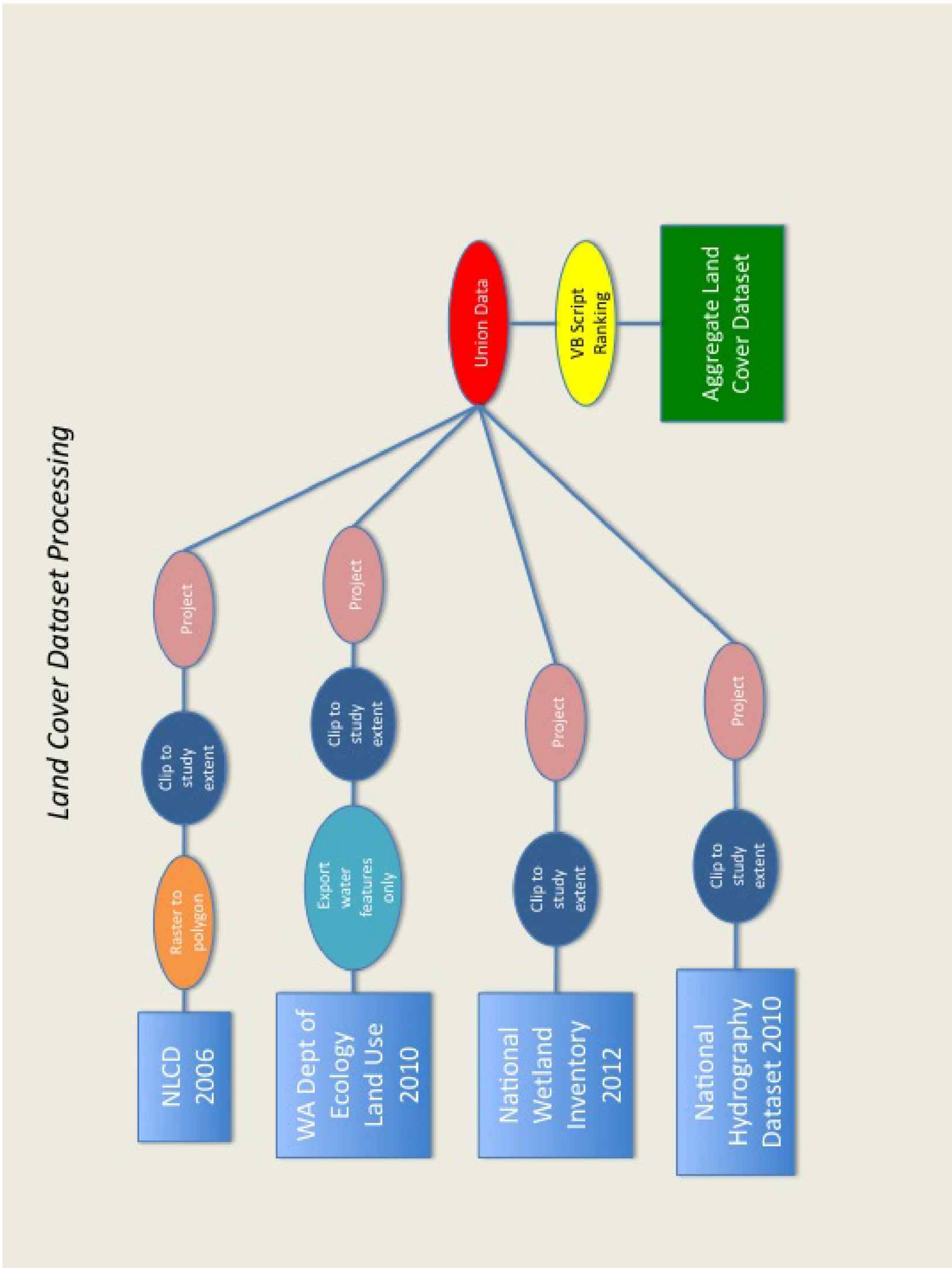
Before beginning the land cover analysis, the extent of the study area required definition. While the Washington State Department of Ecology Water Resource Inventory Area (WRIA) #37, was the expected basis for this analysis, the areal extent the WRIA boundary differed from that of the United States Geological Survey Hydrologic Unit maps for the same region. Because the hydrologic unit codes (HUC) were to be used in subsequent analysis, only the overlapping area in both the WRIA and HUC was used as the extent of the study site. The resulting area was 3,796 hectares (0.5%) smaller than WRIA 37, with area reductions occurring mostly along the eastern border of the study site.

For the land cover analysis, the four datasets used were the National Wetland Inventory (vector, 2012), Washington State Department of Ecology Land Use (vector, 2010), National Hydrography Dataset (vector, 2012) and the National Land Cover Dataset (30-meter raster, 2006). Because of the importance of aquatic ecosystems in the movement of reactive nitrogen, datasets that provided greater precision with regard to water features were preferred over datasets that lacked precision in the description and detail of water features in the LYRB. Particularly given the high level of hydrologic engineering, primarily in the form of irrigation systems, many datasets did not completely inventory the canals and drainages in the study site. After a thorough comparative analysis of datasets, including comparison with irrigation system maps for the area, a ranking system was used to create an aggregated dataset. During the data union process, datasets were given the following ranking:

1. National Wetland Inventory (NWI)
2. Washington State Department of Ecology Land Use (WALU)
3. National Hydrography Dataset (NHD)
4. National Land Cover Dataset (NLCD, Fry et al. 2006)

Before the data were transformed into a single dataset, all datasets were all converted to vector (polygon) format, correcting for differences in projection. The WALU dataset was edited to only include categories of water features as the majority of categories in this dataset were for uses of developed land parcels. After completing a union operation with the datasets, a new field was created using a Visual Basic script in ArcMap field calculator to populate a land cover description field using the above ranking. This field was then used to create a new land cover dataset using ranked land cover typologies from each dataset. A flow chart for this data processing is shown in Figure 4. The aggregate dataset created served as the basis for the land cover analysis and subsequent assessment of ecosystem services.

Figure 4: Land Cover Dataset Processing.



Methods Section 3.2: Benefit Transfer & Ecosystem Service Valuation

Using the land cover analysis as the basis for identifying and quantifying ecosystem services in the LYRB, a benefit transfer approach was used to estimate the value of these services similar to other spatially-explicit ecosystem service valuation studies (Troy and Wilson 2006, Costanza et al. 1997, Eade and Moran 1996, Liu et al. 2010, Kreuter et al. 2001, Schmidt et al. 2011, Schmidt and Batker 2012, Noel et al. 2009). Because the goals of this research were to generate a preliminary ecosystem service valuation that would incorporate an analysis of the relationship between ecosystem services and estimated landscape nitrogen loading, a benefit transfer approach offered an acceptable degree of general valuation focused more on understanding relative value relationships than the generation of the most precise valuation data possible. Economic benefit transfer offers an alternative to extensive primary valuation of nonmarket goods and services that use methods such as hedonic pricing or contingent valuation, which were beyond the given time and funding constraints for this project.

Defined as the "application of values and other information from a 'study' site where data are collected to a 'policy' site with little or no data" benefit transfer is a widely used method of valuation for preliminary value estimates or when conducting primary valuation studies is impractical or impossible (Rosenberger and Loomis 2000). Because of the broad goals of this study to conduct general analysis the benefit transfer approach was by far the most feasible valuation option. While benefit transfer may be completed in the form of a benefit function transfer (where site specific variables are used in the value equation) or a point benefit transfer (where specific values are applied from the 'study' site to the 'policy' site), after an evaluation of existing literature and valuation studies, point benefit transfer was determined to be the appropriate and feasible methodology for this research. Although a benefit function transfer has been argued to increase the validity and reliability of a benefit transfer (Rosenberger and Stanley 2006, Plummer 2009), suitable functions were not found, and given the goals of this research, point estimates of value were judged to be adequate for this analysis.

In search of appropriate studies to use as the basis for benefit transfer, both primary ESV studies and ESV studies using benefit transfer and meta-analysis were reviewed. While a comprehensive review of primary ESV studies, or a complete meta-analysis is beyond the scope of this research, through the review process, it was determined that it would be preferred to identify studies that performed a regional land cover-based ecosystem service valuation and select the valuation data from a study site which would be of best correspondence rather than use individual valuation studies for each of the land cover types present in the LYRB. One reason for this decision was that there was not found to be an abundance of primary valuation studies on the specific land cover types in the LYRB. It could be suggested that the lack of specific land cover ecosystem service valuation studies this research discovered in the literature may be the result of publication goals to demonstrate only the most current valuation techniques rather than simply publishing quality valuation studies using established methods.

After a literature review, several of the studies identified for use in benefit transfer conducted meta-analysis for ecosystem service values based on land cover typology (Troy and Wilson 2006, Costanza et al. 1997, Liu et al. 2010, Noel et al. 2009), while others used the independently developed, analytical valuation software, SERVES (Schmidt et al. 2011, Schmidt and Batker 2012). In meta-analysis, a collection of primary valuation studies from a variety of sources are statistically analyzed to extract relevant environmental values from the outcomes of these previous studies. It is presumed that an underlying valuation equation exists that can be linked to the specific characteristics of the study site (Rosenberger and Stanley 2006). By using meta-analysis values, rather than a simple point value from a single study, it is hoped this will yield greater correspondence between the study and policy sites and minimize generalization error. The generalization error of applying values from one site to another without adequately accounting for differences in site characteristics has been recognized as a significant issue in the application of benefit transfer, and is a critical component to this valuation method (Plummer 2009, Rosenberger and Stanley 2006, Johnston and Rosenberger 2010, Brouwer 2000, Boyle and Bergstrom 1992). To attempt to minimize generalization error, only studies with well-documented valuation and meta-analysis, and common site characteristics were selected for the benefit transfer to estimate ecosystem service values in the LYRB.

Having identified suitable studies to potentially use as the basis for a benefit transfer valuation of ecosystem services in the LYRB, each study was reviewed to determine its suitability for this valuation. While several studies had specific components that demonstrated a high level of correspondence to the LYRB, all studies had significant differences or omissions that limited their suitability for a direct benefit transfer. Studies were either performed in study sites with significantly different ecology (Schmidt et al. 2011, Schmidt and

Batker 2012, Troy and Wilson 2006, Liu et al. 2010), contained enormous differences in study scale (Costanza et al. 1997, Liu et al. 2010) or omitted critical land cover types such as shrub/scrub which is dominant in the LYRB (Schmidt et al. 2011 & Schmidt and Batker 2012 were the only studies to include this land cover type). Because of these differences, a different approach was taken using all of the identified studies to calculate general values for each land cover type base on minimum, maximum, median and average value from all of the studies identified. While this undoubtedly increased generalization error by using studies with large differences from the LYRB geographic and ecological context, patterns in the relative values between different land cover types between different studies can still be sufficiently demonstrated. This approach did not increase the precision of the valuation results, but as a preliminary valuation it is still successful in highlighting differences in ecosystem service values between different land cover types which informed other components of this research.

Methods Section 3.3: Landscape Nitrogen Loading

In order to understand the relationship between nitrogen and ecosystem services in the LYRB spatial data estimates of nitrogen (N) loading were used to examine the distribution and magnitude of N loading. By understanding the characteristics and sources of N loading, it is possible to analyze how this may impact the ecosystem service values provided by benefit transfer. Given the complexity of reactive nitrogen in the environment as previously described, the N loading estimates simply provide a reference for understanding where N is applied, but the loading data do not imply the location or timing of transformation processes such as denitrification. While these processes may be significant in accurately assessing environmental and human health impacts, advanced methods such as isotope analysis would be needed to determine precise movements of Nr in the environment. Still even without detailed data on Nr fate and transport, landscape N loading data provided a useful starting point for this analysis to explore this data along with ecosystem service values, hydrologic boundaries and groundwater data to begin to highlight areas of potential concern due to the location or magnitude of N loading.

The N loading data used in this research was provided by Dan Wise, USGS Water Science Center Portland, Oregon and was generated using regional land use data, the SPAtially Referenced Regression on Watershed attributes (SPARROW) model and the Community Multiscale Air Quality (CMAQ) model. The original datasets contained nine data layers of varying resolution. A description of each data layer and details of how estimates were calculated is shown below (Wise 2011):

<u>Confined Animal Feeding Operations:</u>

60 meter grid of nitrogen loading from waste generated at dairies and feedlots that has been distributed to agricultural land based on proximity (2002).

<u>Farm:</u>

30 meter grid of landscape nitrogen loadings from farm fertilizer use (2002).

Estimates were based on the assumption that the annual amount of farm fertilizer used in a county was applied equally to all cultivated crops and pasture land (NLCD 81 and 82) located within that county.

Nonfarm:

30 meter grid of landscape nitrogen loadings from nonfarm fertilizer use (2002).

Estimates were based on the assumption that the annual amount of nonfarm fertilizer used in a county was applied equally to all nonroad developed land (NLCD 21-24) located within that county.

Nonsewer:

30 meter grid of landscape nitrogen produced by nonsewered populations (2001).

Estimates were based on the assumption that the population within each census block was evenly distributed to all nonroad developed land within that census block. Population located within sewered areas were removed.

Pasture:

30 meter grid of landscape nitrogen loadings from noncattle livestock (2002).

Estimates were based on the assumption that the annual amount of noncattle livestock waste generated in a county (excluding poultry waste) was applied equally to all pasture land (NLCD 82) located within that county.

Range:

30 meter grid of landscape nitrogen loadings from rangeland cattle (2002).

These loadings are from cattle not associated with a dairy or feedlot. Estimates were based on the assumption that the annual amount of rangeland waste generated in a county was applied equally to all potential rangeland (NLCD 41, 42, 43, 52, 71, 81, 90, 95 and accounting for slope, proximity to water, canopy cover and BLM allotments) located within that county.

Atmospheric:

120 meter grid of total atmospheric nitrogen deposition (dry and wet, oxidized and reduced) predicted by the CEMAQ model (2002).

Two additional layers were provided, but not used in the analysis. These layers were:

<u>Crop:</u>

56 meter grid of landscape nitrogen loadings from farm fertilizer use (199?).

Estimates were based on crop-specific nitrogen fertilizer application rates (Washington State University) for crops contained in the USDA Crop Data Layer for 199?.

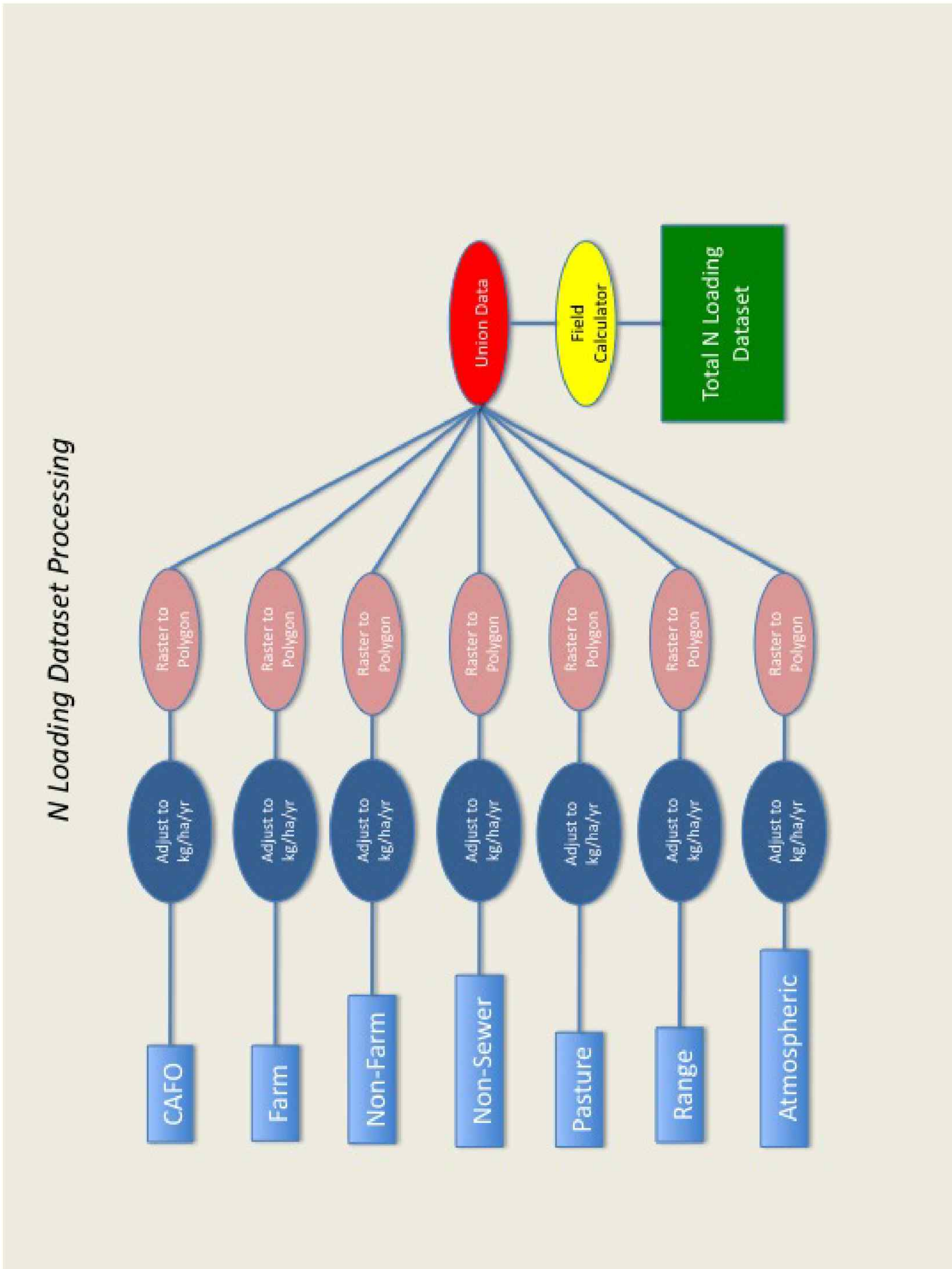
<u>Land:</u>

Location of permitted wastewater facilities that land apply wastewater and nitrogen loading associated with each facility (2002).

The Crop layer provided duplicated fertilizer estimates contained in the Farm layer, and was not included in analysis. The Land layer, while a relevant contributor to N loading in the LYRB, consisted solely of point locations of wastewater facilities, but did not make any estimates of the spatial extent of N loading surrounding these locations. Because of this, land applied N loading data from wastewater facilities was not able to be included as a part of this research.

Due to differences in resolution, each of the seven data layers required individual calculations to convert kg/year load estimates into kg/hectare/year prior to calculating total N load estimates. After this areal conversion, each data layer was then converted into polygon format to allow the union of land cover data and N loading data for subsequent analysis. The aggregated N loading dataset created contained kg/hectare/year information by source for each unique polygon, with total loading rate and annual load by source and total subsequently calculated in ArcMap. The general processing of data layers in represented in Figure 5.

Figure 5: N Loading Dataset Processing



Methods Section 3.4: Estimating Groundwater Ecosystem Services

During the literature review for the benefit transfer component of this research, it became apparent that ecosystem services from subsurface processes and resources were either omitted completely within the land cover based ecosystem service valuation literature, or at best, implied in meta-analysis. With no specific details regarding the values of groundwater ecosystem services, the spatially explicit ecosystem service valuation literature did not offer any opportunity to examine how nitrogen loading impacts on these resources may compare to other ecosystem services in the LYRB. Additionally, because groundwater quality issues related to elevated nitrate levels in the LYRB have been the subject of a great deal of public concern and research (Sell and Knutson 2002, WA DOE 2010, EPA 2012), incorporating the value of these services was a clear priority in this research.

Hoping to identify a suitable methodology for incorporating groundwater values with ecosystem services and nitrogen loading data, the general non-market valuation literature specific to groundwater and groundwater quality was surveyed. While valuation literature on groundwater resources outside of the ecosystem service framework is extensive (NRC 1997, VandenBerg et al. 1995, Poe and Bishop 1999, Poe 1998, Lewandowski 2008, Poe 1997; including meta analysis of contingent valuation studies: Boyle et al 1994), after a review of this literature it was again determined that a benefit transfer approach would be the most suitable for this research. However, unlike the ecosystem service valuation which utilized a point transfer using values from multiple study sites, a suitable function for value transfer was identified from Crutchfield et al. 1997. In Crutchfield et al. 1997, the Mid-Columbia Basin (which encompasses the LYRB study site) was used as the basis for a contingent valuation study that sought to assess the value of nitrate reduction in groundwater. Despite the fact that the total value of groundwater resources is different from simply the value of reducing nitrate levels in groundwater, because the focus of this research is on understanding how nitrogen loading may affect ecosystem service values, the Crutchfield study provided an excellent opportunity to conduct benefit transfer on the specific groundwater characteristic (nitrate level) that is the primary focus of this research. However, it is important to note that the benefit transfer completed using the Crutchfield study is not an estimate of total resource value, like was estimated in the land cover ecosystem service benefit transfer, but is an estimate specifically of the value of reducing nitrate levels in groundwater and therefore is likely a conservative estimate of total groundwater resource value. In terms of ecosystem services, using the Millennium Ecosystem Service categorization, this benefit function provided an estimate of the regulating services that contribute to the reduction of nitrate levels in groundwater, and the provisioning service of groundwater availability at or below the US EPA maximum contaminant level (MCL) for nitrate-N.

Using the results of the contingent valuation study in the Mid-Columbia Basin, the Crutchfield et al. 1997 study value function was constructed using a bivariate probit estimation regression model. The variables used in this regression were the following:

- Personal Annual Income
- Extra Income (Household Income-Personal Income)
- Awareness of nitrate contamination
- Connection to municipal water supply
- Use of a water treatment system
- Purchases bottled water
- Years living at zip code
- Residing in rural area
- Age
- Sex
- Education

While the Crutchfield study provided 1997 mean values for the Mid-Columbia Basin, updated 2010 US Census data were used to estimate variable values on a census tract scale if possible. If updated data were not available, 1997 mean values or estimated values based on regional characteristics were used as described in the groundwater valuation results. The value estimates produced indicate willingness to pay (WTP) for safe drinking water, defined as equal or less than the current EPA drinking water standard for nitrate-N levels at 10 mg/L. These WTP estimates were calculated for each census tract and combined to estimate total WTP for "safe" nitrate levels for the LYRB study site.

Results Section 4.1: Land Cover Analysis

After the data aggregation into a single dataset, the study area was found to contain 37 different land cover classifications from the four datasets used (see Table 2). Because this level of detail is not reflected in the ecosystem service valuation literature, land cover types needed to be generalized to correspond to the level of precision used in valuation studies. As a first step the 27 land cover types were simplified to 12 NLCD land cover classifications using the hierarchy shown in Table 2. Ten of the land cover types associated with developed land or constructed infrastructure were assigned to a category of no ESV (see discussion for more information). Unfortunately, even the precision of the NLCD typology was still too detailed to correspond to all of the valuation studies used as the basis for benefit transfer. These classifications were further generalized using a simplified land classification (Schmidt and Batker 2012, Schmidt et al. 2011). A summary chart of the simplified land cover is shown in Figure 6, with a map of the final simplified land cover in the LYRB shown in Figure 7.

During the land cover typology simplification process, there were decisions that needed to be made regarding land cover type descriptions and ecosystem service values. Specifically, land cover types to be designated as having no ecosystem service value needed to be determined. After reviewing the spatial data the following land cover types were designated as having no ecosystem service value (% of LYRB area shown):

- Canal/Ditch: Aqueduct (0.00003%)
- Canal/Ditch (0.03%)
- Reservoir; sewage treatment pond (0.001%)
- Reservoir; settling pond (0.001%)
- Reservoir; filtration pond (0.001%)
- Reservoir; nonearthen, covered (0.00003%)
- Developed; medium intensity (0.9%)
- Developed; low intensity (2.6%)
- Developed; high intensity (0.1%)
- Barren Land (0.02%)

Together these areas combine for a total of 3.6% of the total area in the LYRB, designated as having no ecosystem service value. While some of these land covers, such as “Canal/Ditch” or “Developed: low intensity” may actually provide ecosystem services, because of the difficulty in assessing this without site specific research, and the relatively small areas represented by these areas, they were assumed to provide a minimal contribution to the total ecosystem service values for the entire study area. Other land cover types in this category, such as “Reservoir: settling pond” may also provide ecosystem services, but clearly this is a land feature that has required a human capital investment and should be considered an engineered ecosystem benefit, using the framework outlined in Section 2.2. Without being able to account for this capital investment, and given the relatively small area covered by these features, these areas were also designated as having no ecosystem service value in order to remain conservative on total ecosystem service value estimates.

Another particular land cover type that required additional categorization was that of “urban green space.” Initially, NLCD 21, Developed, Open Space, was anticipated to represent this category, however upon inspection, many non-urban features, such as open highways, were given this designation. In order to restrict this to urban areas, this land cover type was clipped to only those areas located within city boundaries as provided by Washington State Department of Ecology 2010 datasets (WA-DOE 2010). Areas outside of city boundaries were designated as no ecosystem service value, due to their developed characteristics. This operation resulted in urban green space representing 0.5% of the total LYRB area, as opposed to Developed, Open Land, which originally represented 3.1% of the total area and the NO ESV area increasing from 3.6% to 6.3%.

Using any of the land classification systems (aggregated, NLCD or Simplified) the top three land cover types of shrub/scrub, cultivated crops and evergreen forest represent 384,169 hectares (51.2%), 151,036 hectares (20.1%) and 69,687 hectares (9.3%). Together these three land covers represent approximately 81% of the total land cover in the study area. Land cover associated with aquatic ecosystems [wetlands (2.4%), riverine (0.6%), freshwater pond/lake (0.2%), other water (0.002%)] together comprise approximately 3.2% of LYRB land cover.

Table 2 – Land Cover Typology Hierarchy

Aggregated Data Land Cover Classifications	Area (hectares)	% of LYRB Area	Data Source	NLCD Classification	Area (hectares)	% of LYRB Area	Simplified Land Cover Type	Area (hectares)	% of LYRB Area
Freshwater Pond	713.35	0.1%	NWI	freshwater pond, lake	1697.90	0.2%	Open Water	6133.79	0.8%
Lake	769.05	0.1%	NWI						
Lake/Pond; Intermittent	78.47	0.01%	NHD						
Lake/Pond; Perennial	137.04	0.02%	NHD						
Lake/Pond; Perennial avg level	0.90	0.0001%	NHD	other water	11.48	0.002%			
Other	9.60	0.001%	NWI						
Reservoir; storage	0.98	0.0001%	NHD						
Riverine	2033.36	0.3%	NWI	riverine	4424.40	0.6%			
Stream/River Perennial	322.60	0.04%	NHD						
Open Water	434.38	0.1%	NLCD						
Stream/River	1.41	0.0002%	NHD						
Water areas	1631.49	0.2%	WALU						
Area of Complex Channels	1.15	0.0002%	NHD						
Swamp/Marsh; Perennial	347.04	0.05%	NHD	freshwater forested/shrub wetland, emergent herbaceous wetlands	17972.52	2.4%	Wetland	17972.52	2.4%
Swamp/Marsh; Intermittent	307.47	0.04%	NHD						
Freshwater Forested/Shrub Wetland	3478.32	0.5%	NWI						
Emergent Herbaceous Wetlands	4159.33	0.6%	NLCD						
Freshwater Emergent Wetland	4564.84	0.6%	NWI						
Woody Wetlands	5115.52	0.7%	NLCD						
Developed, Open Space	23461.82	3.1%	NLCD	developed, open space	23461.82	3.1%	Urban Green Space*	3423.32	0.5%
Deciduous Forest	205.31	0.03%	NLCD	deciduous forest	205.31	0.03%	Forest	68195.89	9.14%
Mixed Forest	219.28	0.03%	NLCD	mixed forest	219.28	0.03%			
Evergreen Forest	67771.29	9.1%	NLCD	evergreen forest	67771.29	9.1%			
Pasture/Hay	24093.82	3.2%	NLCD	pasture/hay	24093.82	3.2%	Pasture	24093.82	3.2%
Grassland/Herbaceous	45711.89	6.1%	NLCD	grassland/herbaceous	45711.89	6.1%	Grassland	45711.89	6.1%
Cultivated Crops	150224.87	20.1%	NLCD	cultivated crops	150224.87	20.1%	Agricultural Land	150224.87	20.1%
Shrub/Scrub	383297.93	51.4%	NLCD	shrub/scrub	383297.93	51.4%	Shrub/Scrub	383297.93	51.4%
Canal/Ditch; Aqueduct	0.26	0.00003%	NHD	NO ESV	26764.48	3.6%	NO ESV*	46802.98	6.3%
Canal/Ditch	244.53	0.03%	NHD						
Reservoir; sewage treament pond	5.63	0.001%	NHD						
Reservoir; settling pond	6.10	0.001%	NHD						
Reservoir; filtration pond	6.29	0.001%	NHD						
Reservoir; nonearthen, covered	0.20	0.00003%	NHD						
Developed, Medium Intensity	6368.72	0.9%	NLCD						
Developed, Low Intensity	19084.90	2.6%	NLCD						
Barren Land	178.44	0.02%	NLCD						
Developed, High Intensity	869.40	0.1%	NLCD						

Total LYRB Area (hectares)	745857

Urban Green Space is only Developed, Open Space (NLCD 21) within City limits (Ecology 2012).

** NO ESV increases due to the addition of Developed, Open Space (NLCD 21) outside of city limits*

Figure 6: Simplified Land Cover Summary

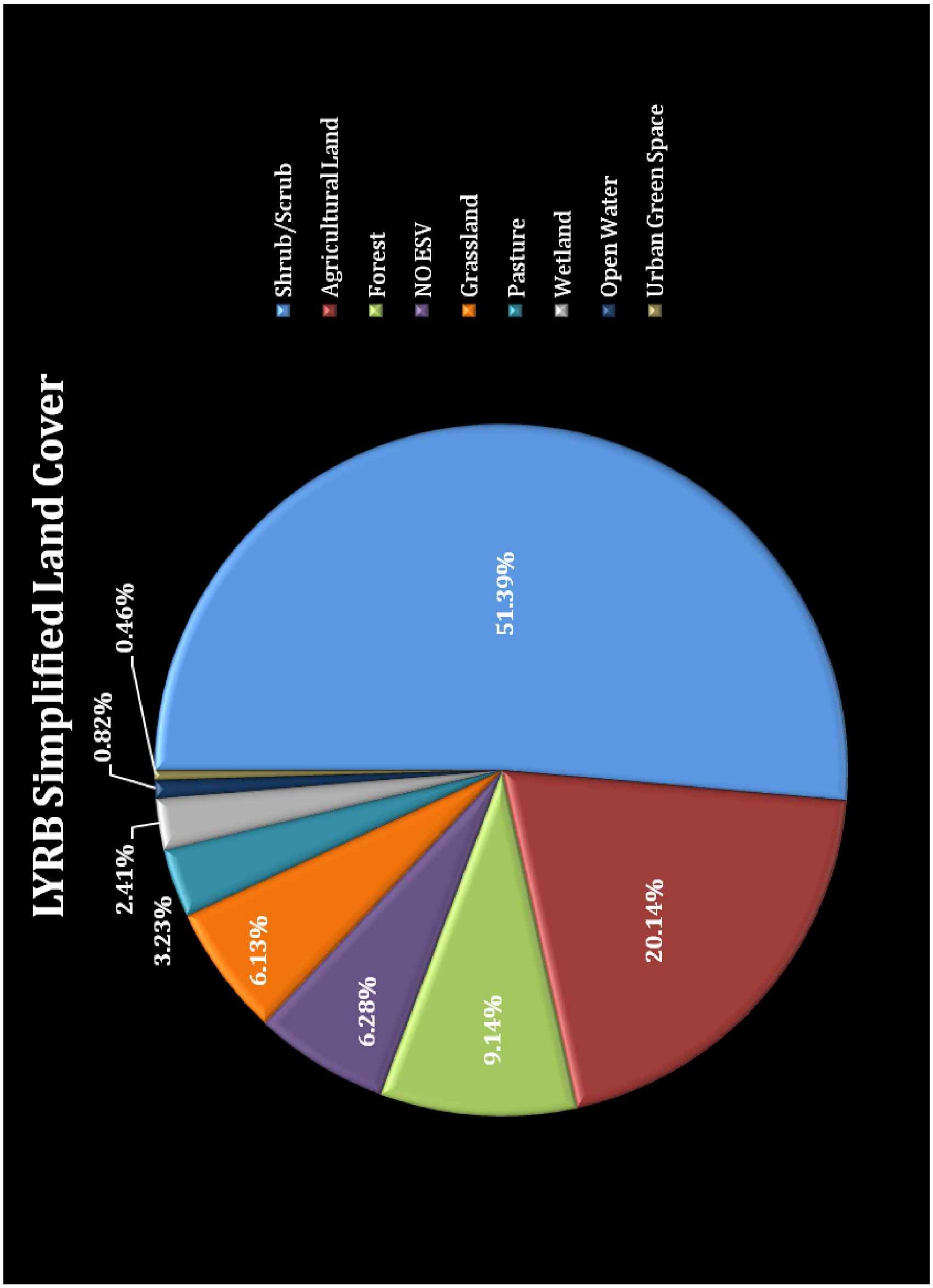
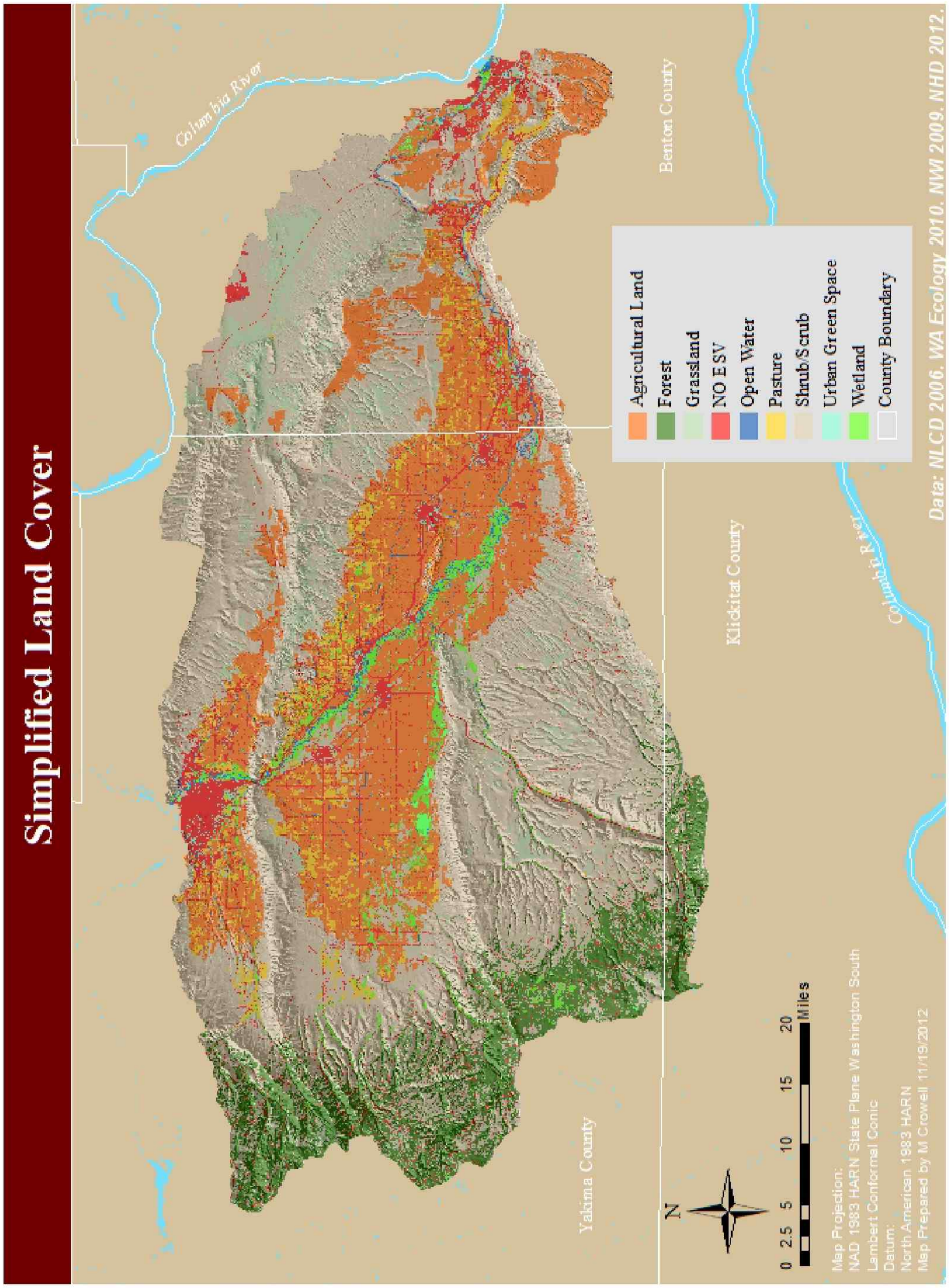


Figure 7: Simplified Land Cover Map



Results Section 4.2: Benefit Transfer Values & Ecosystem Service Valuation

In analyzing studies for the benefit transfer, it became evident that the level of correspondence between any given study and the LYRB was low. None of the studies found included all of the land cover types present in the LYRB, nor did any of the studies adequately reflect the agricultural setting. Because of concerns over this lack of correspondence, rather than select values from one particular study, six studies (Schmidt and Batker 2012, Batker et al. 2011, Troy and Wilson 2006, Liu et al. 2010, Costanza et al. 1997, Noel et al. 2009) were selected to use as the basis of the benefit transfer using combined values. While this approach will also lead to generalization errors, it is expected that by using values across multiple studies the general underlying value relationships between land cover types will be more accurately reflected than if only a single study was selected for benefit transfer. All land cover categories from the studies were converted to \$/hectare/year, adjusted for inflation to 2012 dollars and assembled (see Appendix 1 & 2). During this process several land categories required alterations to fit within the simplified land cover typology used in the land cover analysis. The details of these simplifications are shown in Appendix 2.

After eliminating land cover types not found in the LYRB, studies were reviewed to determine the specific ecosystem services for each land cover type valued. Unfortunately because the definition and accounting framework for ecosystem services is not uniform in the ecosystem service valuation literature different categorizations were used in different studies, with most of the categorizations used similar to those of the Millennium Ecosystem Assessment 2005 (MA 2005). One of the studies used, Troy and Wilson 2006, summarized only specific ecosystem services for one of their study sites (Maury Island, WA), so it is unknown which specific services were included in their valuation of study sites in Massachusetts and Northern California. Using the categorization framework of Schmidt et al. (2011), Schmidt and Batker (2012), specific ecosystem services valued for each land cover type by each study used in this research are shown in Table 3. Analysis of the services included by these studies, and their relevance to the LYRB can be found in Discussion Section 5.2.

From the values provided by each study, the average, median, minimum and maximum ecosystem service benefit flow rate (\$/ha/yr) was calculated for each simplified land cover type as shown in Table 4. Using these ecosystem service value rates and the area of each land cover type, the annual value of ecosystem services for the entire LYRB and for each land cover type was calculated (Table 5). Because the minimum and maximum values both showed considerable variability influenced by specific studies, these extreme values were omitted from subsequent analysis due to the fact they represented a more dramatically conservative or liberal estimation. Average value estimates also appeared to be influenced by minimum and maximum values, and therefore the median estimates (high/low) were selected as the basis for further analysis. Using these values, the total annual value of ecosystem services can be estimated at between $554,320,594 and $3,996,211,889 per year. While these numbers demonstrate a very wide variation, they are intended to serve as preliminary estimates to provide a general level of estimation of ecosystem services. Because ecosystem service values vary significantly based upon how benefits are received and the actual productivity and health of an ecosystem for any land cover, it is expected that high/low estimates would cover a significant range. Despite this wide range, the contribution of each land cover type to the annual ecosystem service value is still a critical component for understanding the role of ecosystem services in this area and these contributions are shown for low/high median estimates in Figures 8 and 9.

Because the distribution of ecosystem service values is also critical to the analysis of the impacts of nitrogen loading, ecosystem service values were combined with land cover GIS data to map ecosystem service values in the LYRB, shown in Figures 8 and 9. Both of these figures clearly show the importance of rivers, wetlands and riparian zones in terms of ecosystem service values. Using the high median estimates, contributions from shrub/scrub land cover become more significant, however the clustering of high ecosystem service values along the Yakima River and significant tributaries remains clear. In the low median estimate it is also clear that forest land cover along the western edge of the study site also appear to contain higher ecosystem service values within a clustered region.

In terms of land cover types used for this analysis, wetlands were found to have significantly higher value per hectare than other land cover types. However, because of differences in high and low per hectare estimates, wetland land cover is only the largest contributor to total ecosystem service values using the low estimates. With the high estimations, shrub/scrub land cover becomes the dominant contributor, largely due to the large area of this land cover in the LYRB. Median ecosystem service flows for both high and low estimates clearly indicate that wetland land cover types are found to have a much higher value than any other land cover type found in the LYRB. This is further demonstrated in Figures 10 and 11, where total ecosystem service value and total area by land cover type are shown together. Examining the graphs of high and low ecosystem service value estimates and area for each land cover type, clearly illustrate the disproportionately high contribution of wetlands and the importance of the ecosystem service value of shrub/scrub land cover because of its large area in the LYRB.

Table 3: Ecosystem Services Valued

	Agricultural Lands	Forest	Grasslands	Lakes/Rivers	Pasture	Shrub/Scrub	Urban Green Space	Wetland
Provisioning								
Water Supply		1,2,3,5,6	1	**ALL**		5		1,2,3,5,6
Food, Food Production	3	3,5,6	3,5,6	3,5				3,5,6
Raw Materials		3,5,6						3,6
Genetic Resources		3,5,6	3					
Medicinal Resources								
Ornamental Resources		6						
Regulating								
Gas Regulation	5,6	1,2,3,5,6	1,2,3,5,6			5,6	2,4,5,6	3,5,6
Climate Regulation	5,6	**ALL**	1,2,3,5,6			5,6	2,4,5,6	5,6
Disturbance Prevention	5,6	3,5,6						1,3,5,6
Soil Retention	5,6	3,5,6	1,3,5,6			5,6	2,4,5,6	
Water Regulation	4	1,3,5,6	1,3,5,6	1,3,4		6		**ALL**
Biological Control	3	3,5,6	3,5,6		5,6	6		
Water Quality, Waste Treatment		3,5,6	1,3,5,6	3		5		1,3,4,5,6
Soil Formation	4	3,5,6	1,2,5,6		5,6	6		
Nutrient Regulation	5,6	3,5,6						6
Pollination	2,3,4,5,6	2,5,6	3,5,6		5,6	5,6		
Habitat								
Habitat and Biodiversity	4	1,2,4,5,6		1,5,6		5,6		**ALL**
Nursery		5,6		5,6		5,6		5,6
Information								
Aesthetic Information	2,4,5,6	1,2,4,5,6	2	1,2,4,5,6	5,6	5,6	2,4,5,6	1,2,4,5,6
Recreation	2	**ALL**	1,2,3	**ALL**	5,6	5,6	2,4,5	**ALL**
Cultural and Artistic Information	4	3				6		3
Science and Education		6		5			5,6	
Spiritual and Historic Information	4							

*1 Troy and Wilson 2006 (*Maury Island WA only)*
2 Liu et al. 2010
3 Costanza et al. 1997
4 Qenani-Petrela et al. 2007
5 Schmidt and Batker 2012
6 Batker et al. 2011

Table 4: Low/High Ecosystem Service Value Rates (2012 USD)

Land Cover Type	LOW ESV ($/ha/yr) avg. value	median	min	max
Agricultural Lands	$1,450.73	$232.17	$69.34	$4,180.94
Forest	$1,381.61	$1,011.10	$623.42	$3,867.84
Grassland	$252.80	$359.60	$36.18	$362.28
NO ESV	$0.00	$0.00	$0.00	$0.00
Open Water	$3,678.62	$1,145.58	$90.24	$13,171.90
Pasture	$1,420.01	$48.71	$48.70	$4,162.64
Shrub/Scrub	$210.96	$210.96	$210.96	$210.96
Urban Green Space	$5,457.33	$5,478.67	$3,746.62	$7,455.32
Wetland	$19,599.87	$18,151.91	$1,783.42	$40,195.34

Land Cover Type	HIGH ESV ($/ha/yr) avg. value	median	min	max
Agricultural Lands	$1,858.13	$1,699.88	$69.34	$4,180.94
Forest	$4,754.00	$3,398.31	$841.09	$14,544.21
Grassland	$836.05	$359.60	$36.18	$1,820.26
NO ESV	$0.00	$0.00	$0.00	$0.00
Open Water	$20,371.63	$9,489.39	$1,945.90	$60,352.52
Pasture	$2,193.65	$1,209.15	$1,209.15	$4,162.64
Shrub/Scrub	$7,098.61	$7,098.61	$7,098.61	$7,098.61
Urban Green Space	$9,726.59	$8,894.97	$6,772.74	$14,175.83
Wetland	$60,624.07	$36,388.22	$13,270.64	$154,024.06

Table 5: Total Ecosystem Service Estimates (2012 USD)

Land Cover Type	LYRB Area	Average ESV ($/yr)		Median ESV ($/yr)	
	hectares	*low*	*high*	*low*	*high*
Agricultural Lands	150224.87	$217,935,487	$279,137,616	$34,877,979	$255,364,197
Forest	68195.89	$94,220,342	$324,203,584	$68,952,654	$231,750,779
Grassland	45711.89	$11,556,183	$38,217,556	$16,437,997	$16,437,997
NO ESV	46802.98	$0	$0	$0	$0
Open Water	6133.79	$22,563,873	$124,955,204	$7,026,743	$58,205,894
Pasture	24093.82	$34,213,569	$52,853,364	$1,173,538	$29,133,102
Shrub/Scrub	383297.93	$80,860,990	$2,720,881,721	$80,860,990	$2,720,881,721
Urban Green Space	3423.32	$18,682,189	$33,297,222	$18,755,233	$30,450,304
Wetland	17972.52	$352,259,017	$1,089,567,192	$326,235,459	$653,987,905
total ESV ($/yr)		**$832,291,651**	**$4,663,113,459**	**$554,320,594**	**$3,996,211,899**

Land Cover Type	LYRB Area	Minimum ESV ($/yr)		Maximum ESV ($/yr)	
	hectares	*low*	*high*	*low*	*high*
Agricultural Lands	150224.87	$10,416,258	$10,416,258	$628,081,181	$628,081,181
Forest	68195.89	$42,514,682	$57,358,882	$263,770,911	$991,855,157
Grassland	45711.89	$1,653,684	$1,653,684	$16,560,332	$83,207,707
NO ESV	46802.98	$0	$0	$0	$0
Open Water	6133.79	$553,499	$11,935,735	$80,793,625	$370,189,486
Pasture	24093.82	$1,173,282	$29,133,102	$100,293,888	$100,293,888
Shrub/Scrub	383297.93	$80,860,990	$2,720,881,721	$80,860,990	$2,720,881,721
Urban Green Space	3423.32	$12,825,881	$23,185,250	$25,521,921	$48,528,353
Wetland	17972.52	$32,052,518	$238,506,800	$722,411,516	$2,768,200,418
total ESV ($/yr)		**$182,050,795**	**$3,093,071,432**	**$1,918,294,364**	**$7,711,237,912**

Figure 8: ESV Contribution by Land Cover (Low Median)

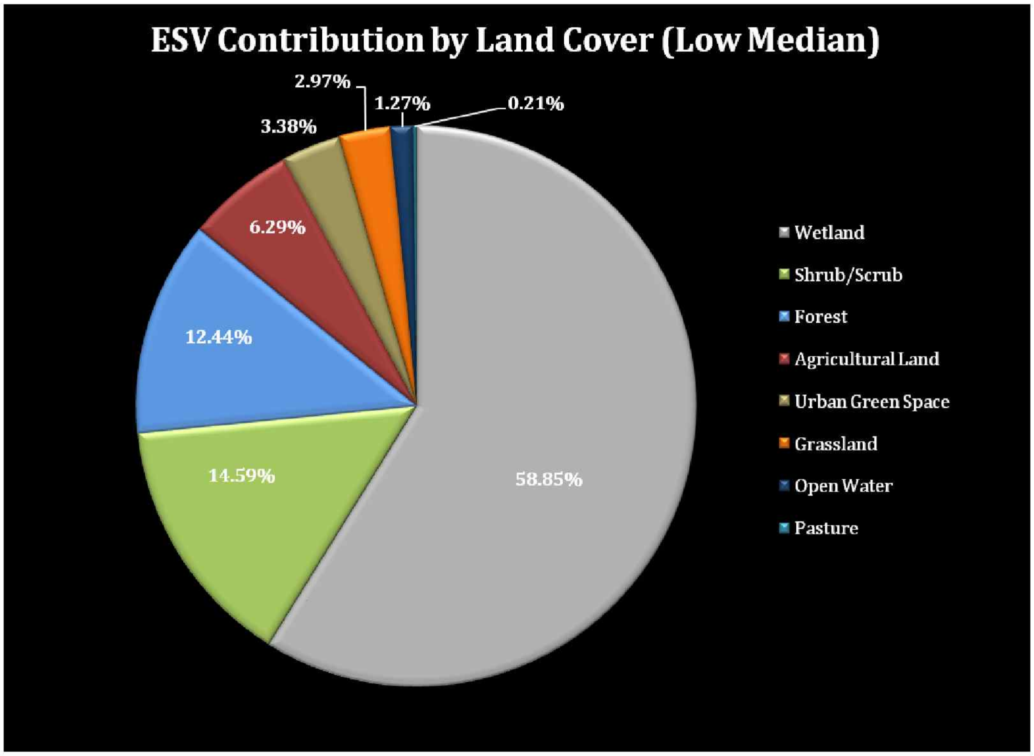


Figure 9: ESV Contribution by Land Cover (High Median)

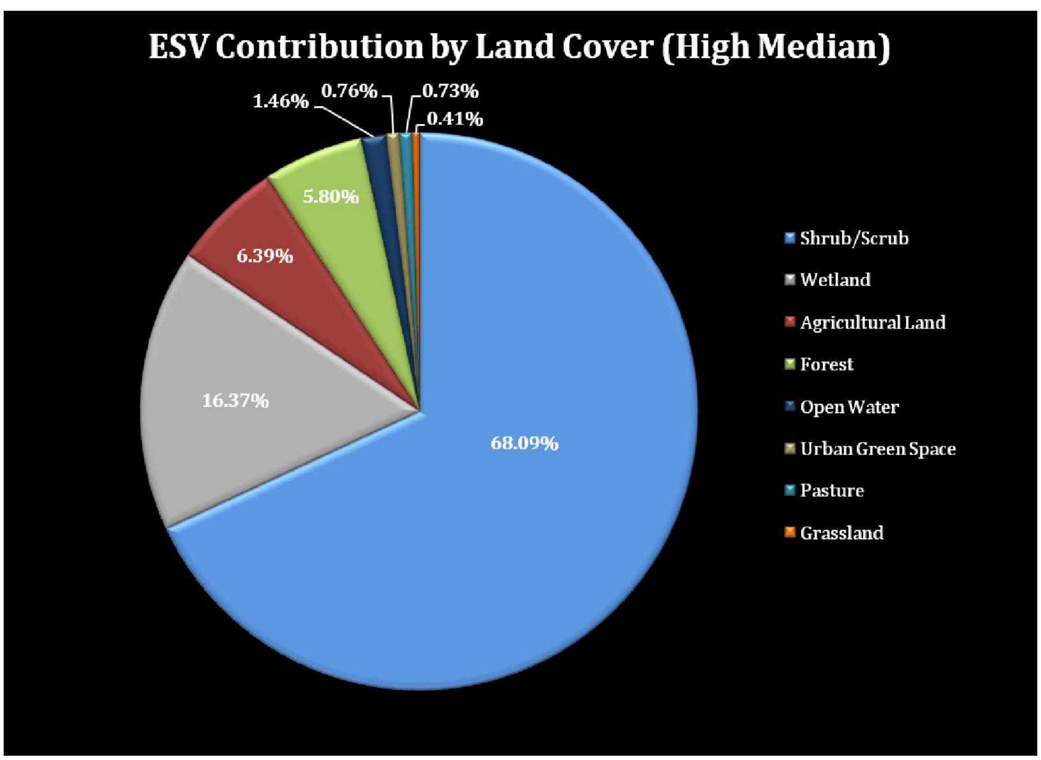


Figure 10: Map of Ecosystem Service Values (Low Median)

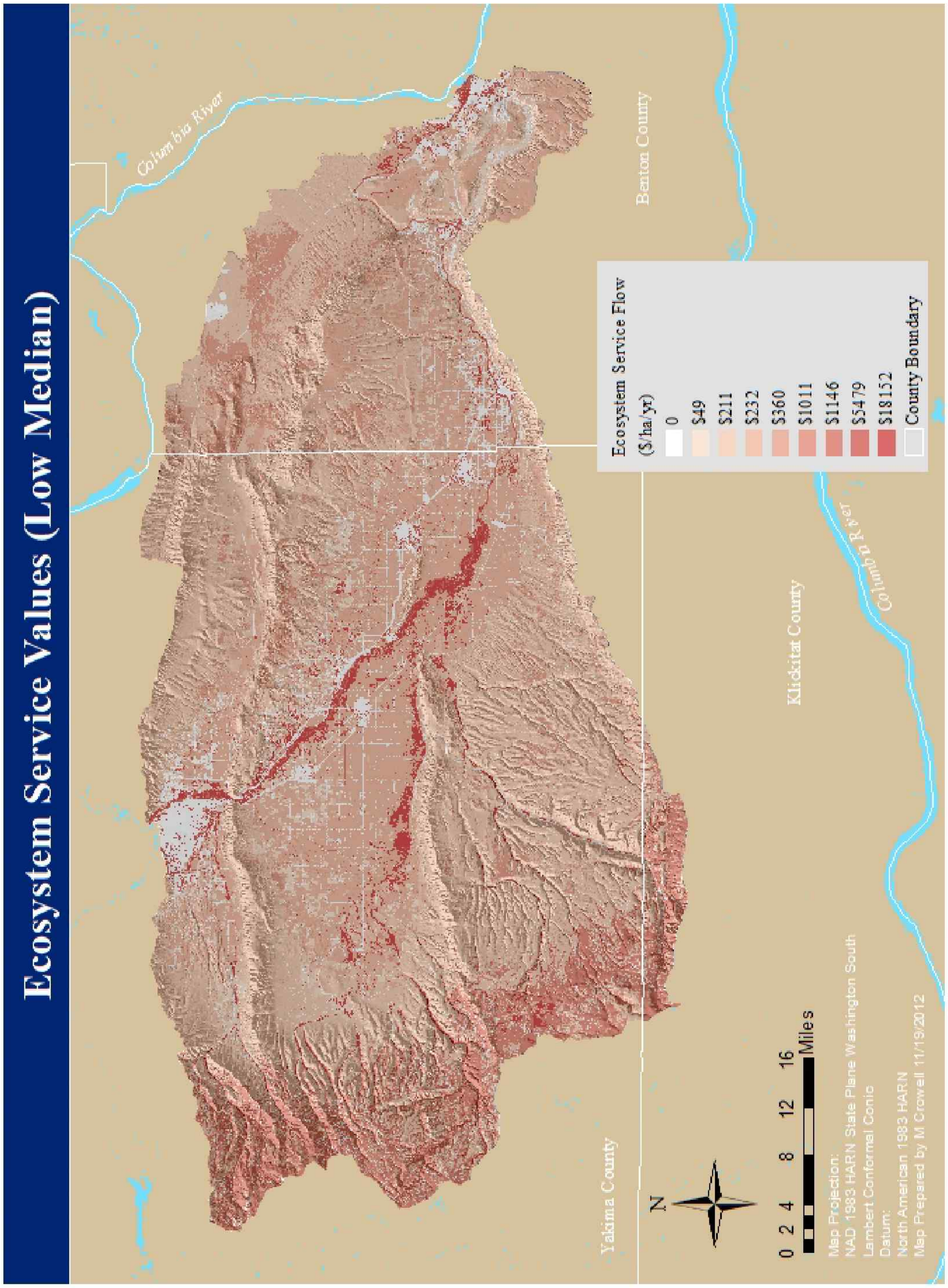


Figure 11: Map of Ecosystem Service Values (High Median)

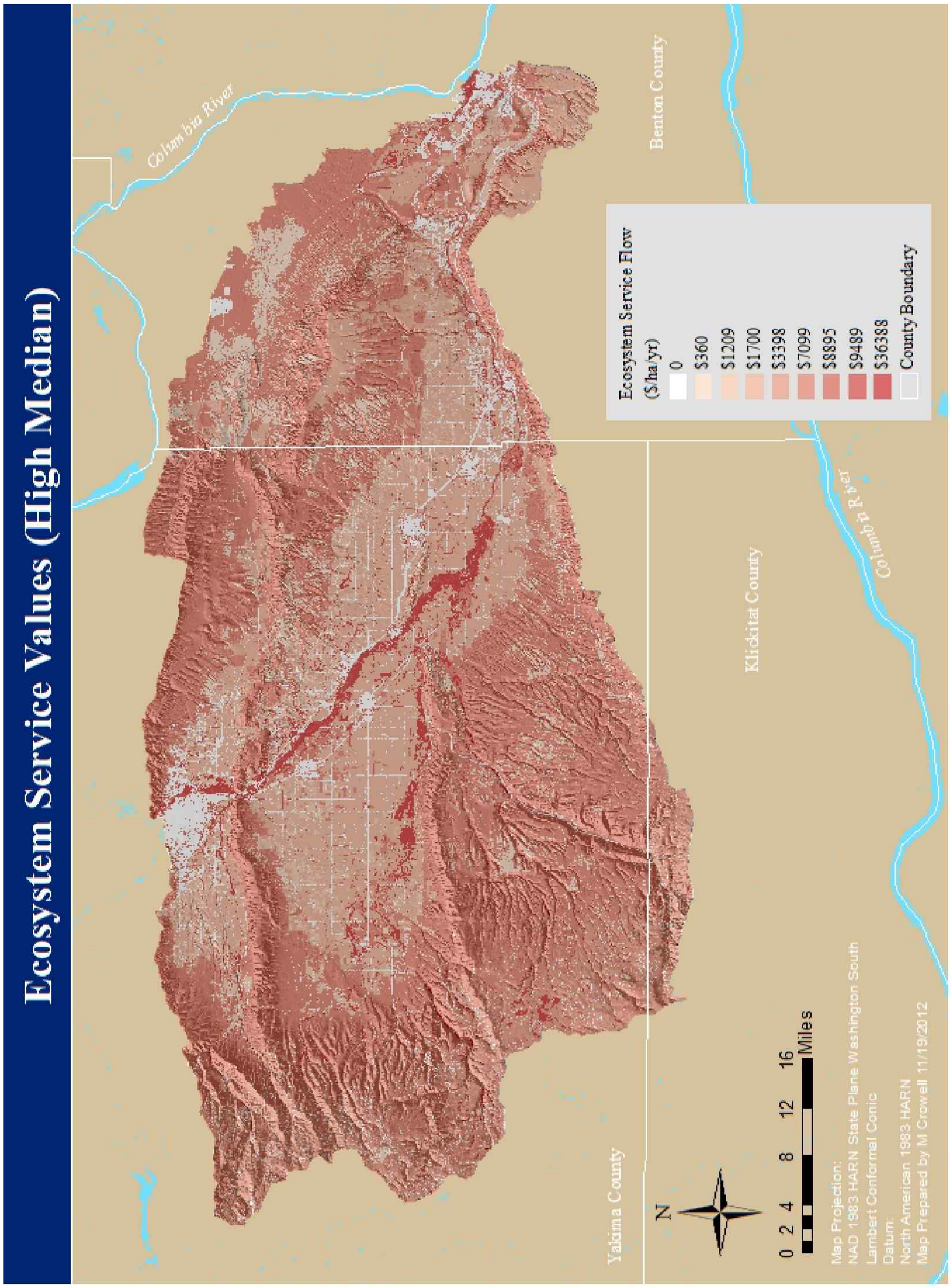


Figure 12: Area and Total ESV by Land Cover Type (Low Median)

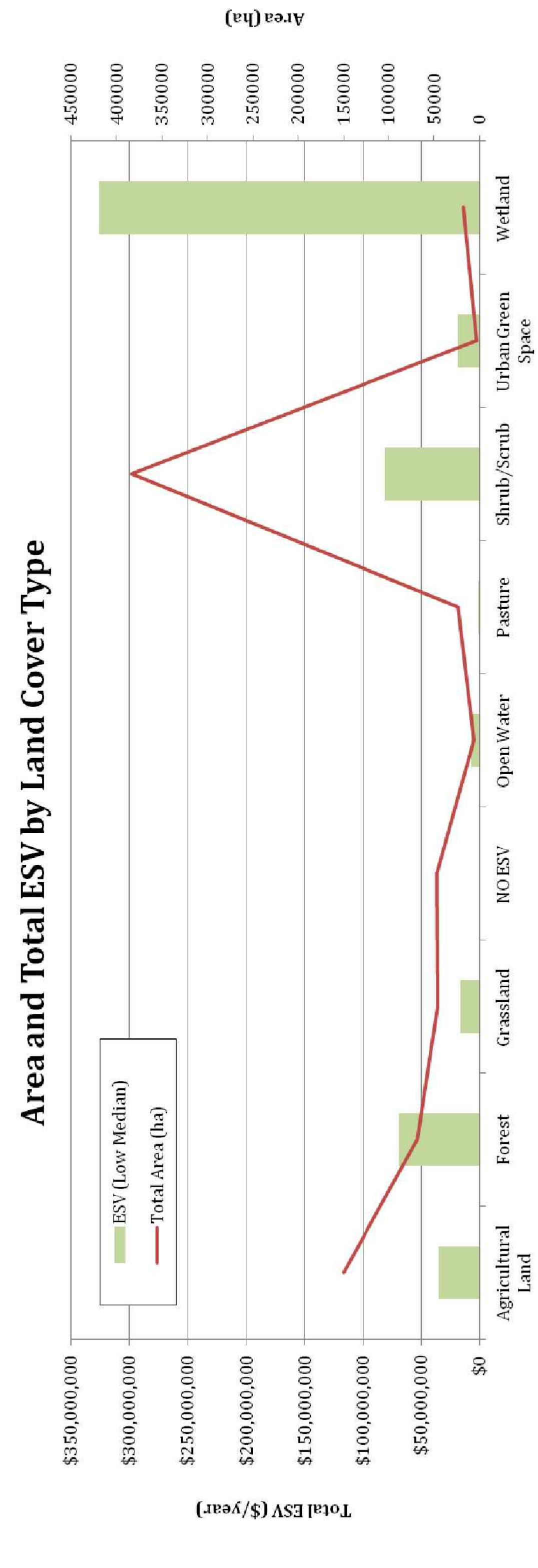
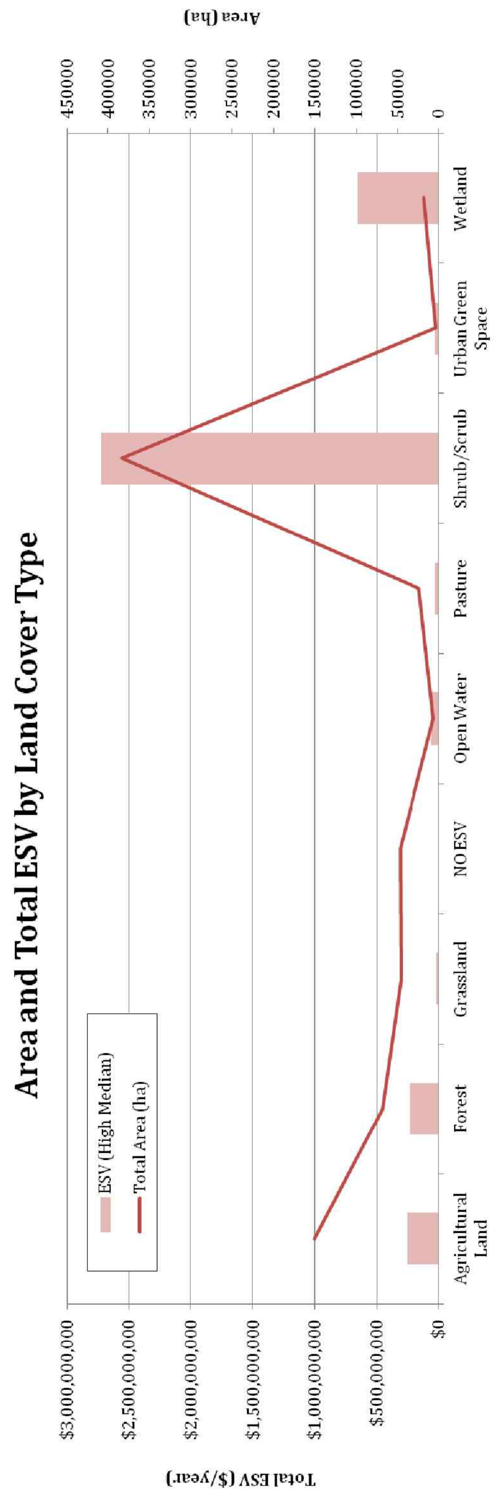


Figure 13: Area and Total ESV by Land Cover Type (High Median)



Results Section 4.3: Nitrogen Loading Data

With ecosystem service values calculated from the benefit transfer the next step was to compile nitrogen loading data to integrate into ArcGIS for spatial analysis with mapped ecosystem service values. Using the methods and data processing described in Methods Section 3.3, a dataset for estimated total annual landscape nitrogen loading (kg/yr) along with estimated average nitrogen loading rates (kg/ha/yr) was created. Average nitrogen loading rates were mapped for the LYRB as shown in Figure 14. In addition to mapping nitrogen loading in the study site, the relationship between land cover types and annual nitrogen load was analyzed as shown in Figure 15. As represented in this figure, agricultural land received dramatically more (73.64% total) nitrogen loading than any other land cover. Other significant results of this analysis indicate pasture land (10.72%) receives the second highest annual nitrogen load (although still significantly less than that of agricultural lands) and that shrub/scrub receive the least nitrogen loading in proportion to area in the LYRB.

The dataset created also permitted an analysis of nitrogen loading sources as shown in Figure 16. From this it is clear that CAFO and farm inputs contribute significantly higher amounts of nitrogen to the study site than any other nitrogen sources. To understand the relationship between nitrogen sources and the total nitrogen load, source loading and total loading at an individual polygon scale was tested for correlation to understand statistically how the level of nitrogen from a given source relates to the total nitrogen load. The Pearson correlation coefficients (r) between total nitrogen load and nitrogen load from a given source are shown below:

Atmospheric: $r = 0.34$
CAFO: $r = 0.98$
Farm: $r = 0.39$
Non-Farm: -0.085
Non-Sewer: 0.005
Pasture: -0.09
Range: -0.12

From these correlation coefficients there are several key results. By far, CAFO nitrogen loading is the strongest and most significant in terms of driving nitrogen loading in a given location in the LYRB. With such a high r-value, it is clear that if CAFO loading is present, it will dominate total loading from all other sources. A similar relationship is seen in farm and atmospheric nitrogen sources, although to a lesser extent and without the same statistical strength. Another interesting result, although not statistically significant, is the negative r-values for non-farm, pasture and range sources. While these values are not strong enough to draw correlation between these sources and total nitrogen loading, the potential for certain nitrogen sources to show a relationship to areas with lower total nitrogen loading was an unanticipated result.

Figure 14: Average N Loading Rates Map

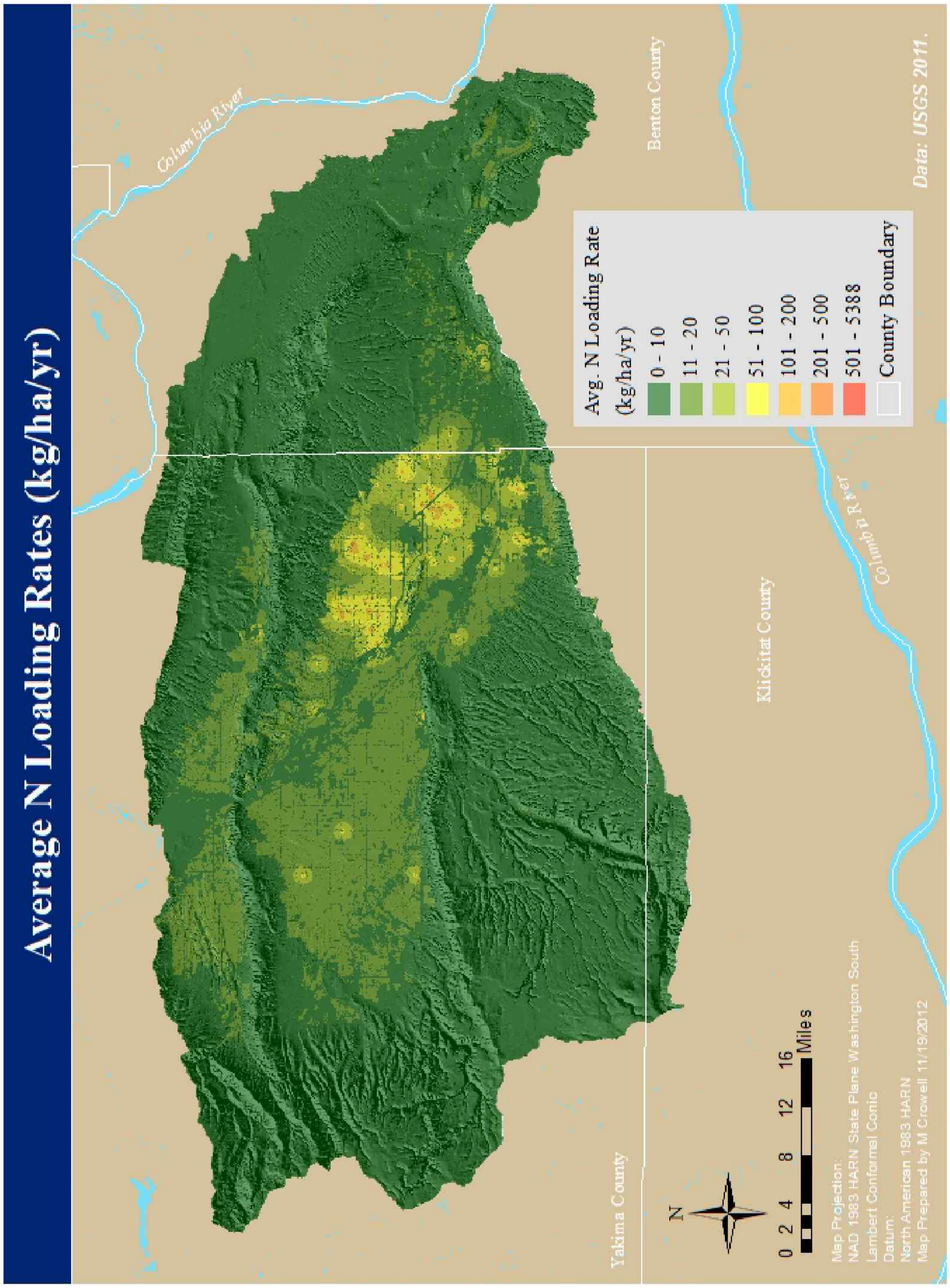


Figure 15: Land Cover Areas and Nitrogen Loads

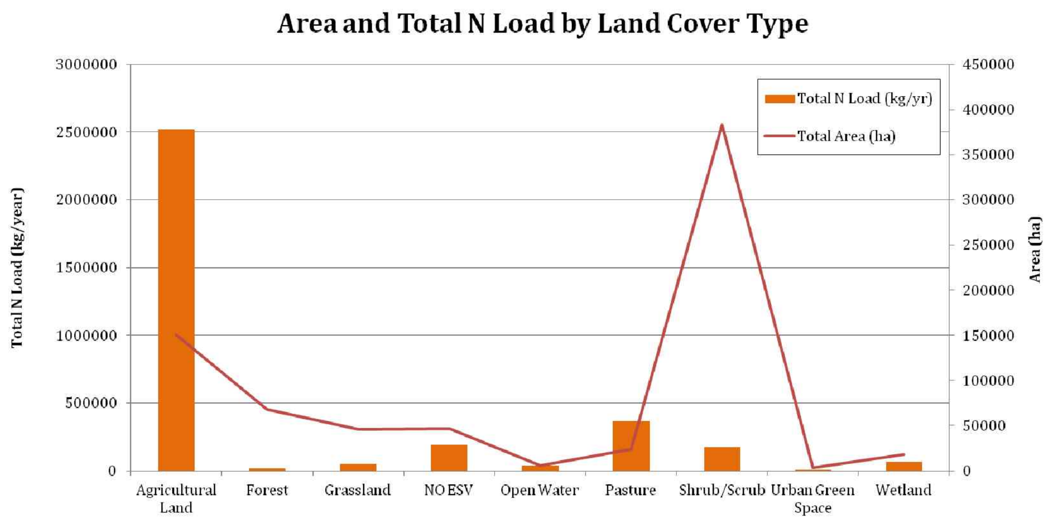
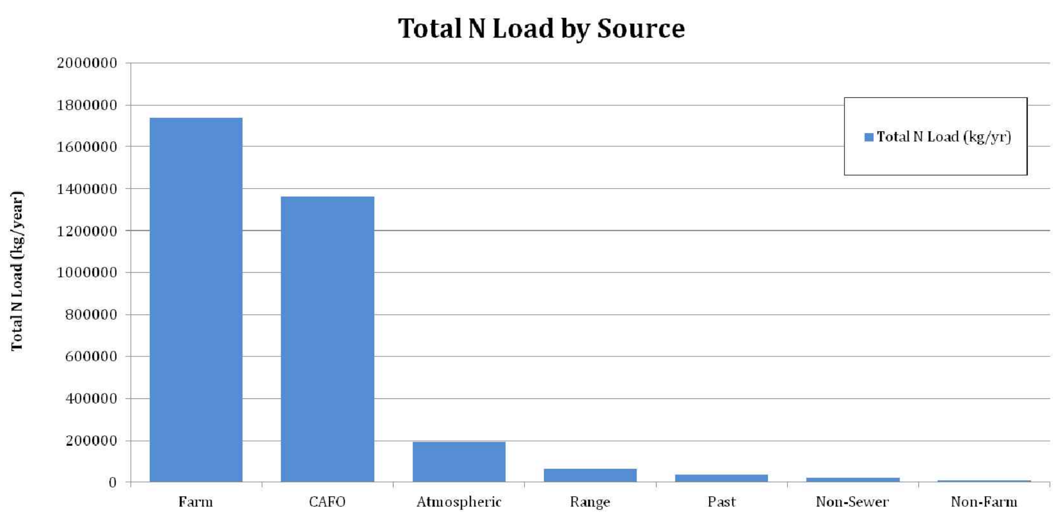


Figure 16: Estimated Nitrogen Load by Source



Results Section 4.4: Nitrogen Loading and Ecosystem Services

Because one of the most important goals of this research was to connect the magnitude and distribution of ecosystem service values with nitrogen loading, several methods of statistical and spatial analysis were performed to understand these relationships. Combining these datasets in ArcGIS provided the opportunity to explore statistical relationships and spatial distributions from a perspective previously undocumented in the ecosystem service literature.

The first step in this analysis compared the total ecosystem service values of land cover types in the LYRB with the annual nitrogen loading delivered. As shown in Figures 17 and 18, using either low or high median estimates of ecosystem service value there is a wide variety of both ecosystem service values and nitrogen loads for each land cover type. At one extreme, wetlands generate the highest (58.85%) annual ecosystem service value using low median estimates, yet receive only a small portion (1.84%) of the total nitrogen load. High median estimates of ecosystem service values become driven by the disproportionately large area of shrub/scrub, but still show similar characteristics for wetlands as low median estimates. Using high or low median estimates, agricultural lands receive a large percentage (73.6%) of total nitrogen but contribute only 6.3-6.4% (low/high) of total ecosystem service value. Table 6 summarizes nitrogen loading and ecosystem service values by land cover and source.

An initial hypothesis of this research was that areas of high ecosystem service value would have a statistical relationship to areas of high nitrogen loading based on the land use patterns and ecology of the LYRB. To test this hypothesis, a correlation test was performed to determine if there was a statistically significant relationship between the nitrogen load in any given polygon and the ecosystem service value estimate for that polygon. This test tried to determine if high nitrogen loading occurred in the same location where high ecosystem service values would be expected. At this scale, it was found that the Pearson correlation coefficient was small ($r = -0.04.$, low median), indicating no correlation, and rejecting the hypothesis that high nitrogen loading and high ecosystem service values occur in the same location.

To refine this hypothesis at a more general scale, USGS hydrologic unit codes (HUC) 10 and 12 were used to again test for correlation between ecosystem service values and nitrogen loading. While the first test indicated that nitrogen loading and ecosystem service values do not occur in the exact *same* location, perhaps there would be a significant correlation at the HUC 10 or HUC 12 level, indicating that they do occur *near* one another at either the watershed or subwatershed scale. Starting at the smaller HUC 12 scale, nitrogen loading data and ecosystem service values were calculated for each of the 72 HUC 12 subwatersheds in the LYRB and the total annual nitrogen load and ecosystem service value were tested for correlation. These values are plotted in Figures 19 and 20 for each HUC 12 with high/low median ecosystem service value estimates. The graphs of ecosystem service values and nitrogen loads at the HUC 12 scale indicate the Pearson's coefficient is small ($r = 0.009$) and there is no correlation between these characteristics. This also rejects the hypothesis that high nitrogen loading and high ecosystem service values occur in the same HUC 12 subwatershed. But while the correlation was still not strong, the r-value did increase from the polygon scale to the HUC 12 subwatershed scale, so the larger HUC 10 scale was also used to test for correlation in the hopes that the strength of correlation would increase significantly at a coarser scale.

Annual nitrogen loads and ecosystem service values were calculated for each of the 12 HUC 10 watersheds in the LYRB as shown in Figures 21 and 22 (low/high median). Despite the increased generalization at the larger HUC 10 scale, the strength of correlation ($r = 0.18$) was still not strong enough to reject the null hypothesis that nitrogen loading does not demonstrate a correlation with ecosystem service values. While the correlation coefficient did increase as the data were tested from the polygon scale to the HUC 12 scale and the HUC 10 scale, there was not enough statistical strength to support that nitrogen loading demonstrates any correlation with ecosystem service values.

Despite not being able to demonstrate a correlation between nitrogen loading and ecosystem service values, there was still ample opportunity for continued analysis using the HUC 12 and HUC 10 data. Because a goal of this research is to identify specific areas or ecosystems services that may be affected by nitrogen loading in the study area, focusing on areas with relatively high ecosystems service values and high nitrogen loading, may provide an opportunity to understand the potential impacts in terms of benefit loss to ecosystem services due to excess nitrogen. After reviewing the distribution of values for HUC 10 and HUC 12 areas, two regions (HUC 10 - Spring Creek; HUC 12 - Horseshoe Lake) were identified as being in the upper quartile at each scale for both ecosystem service value and nitrogen loading. Because of the characteristics of both high nitrogen loads and high ecosystem service values, these areas were selected as the basis for further analysis.

Additionally, due to the policy relevance and high nitrogen loading within the boundaries of the recently established Groundwater Management Area (GWMA) this region was also selected to be included in this analysis.

Similar to the analysis performed for the entire LYRB, nitrogen load by land cover along with land cover area was plotted for each of the areas highlighted, see Figures 23, 25 and 27. Similar to earlier results, nitrogen loading in each of these areas occurs predominately on agricultural lands (GWMA: 77.5%, Spring Creek: 83.3%, Horseshoe Creek: 89.1%). The sources of nitrogen loading in each of these areas are shown in Figures 24, 26 and 28 and clearly indicate the significance of nitrogen loading from CAFO and farm sources in these areas. For a complete summary of nitrogen loading and ecosystem service values in the LYRB and in the GWMA and HUC 10 and 12 study sites see Table 6. Because ecosystem service benefits could only be estimated by the precision of simplified land cover a more detailed analysis could not be conducted, however the nitrogen loads at the more detailed aggregate land cover dataset are shown in Tables 7, 8 and 9. To complement the tabular data on nitrogen loading quantities/sources and ecosystem service values the spatial distributions of average nitrogen loading and ecosystem service values (low median) are shown in Figures 29-34.

Table 6: N Loading Summary

		N Loading Summary							
		HUC 8		GWMA		HUC 10		HUC 12	
		LYRB	% of total	GWMA	% of total	Spring Creek - Yakima River	% of total	Horseshoe Lake - Yakima River	% of total
Land Cover (Hectares)	Agricultural Land	150224.87	20.14%	58937.68	44.75%	33362.95	37.47%	6651.70	42.70%
	Forest	68195.89	9.14%	22.86	0.02%	3.96	0.00%	NA	NA
	Grassland	45711.89	6.13%	5139.73	3.90%	3854.36	4.33%	635.97	4.08%
	NO ESV	46802.98	6.28%	14645.84	11.12%	6693.35	7.52%	772.67	4.96%
	Open Water	6133.79	0.82%	2616.36	1.99%	1603.03	1.80%	356.38	2.29%
	Pasture	24093.82	3.23%	10063.78	7.64%	1900.08	2.13%	33.70	0.22%
	Shrub/Scrub	383297.93	51.39%	35698.80	27.11%	38009.37	42.69%	6195.41	39.77%
	Urban Green Space	3423.32	0.46%	1072.40	0.81%	376.64	0.42%	2.52	0.02%
	Wetland	17972.52	2.41%	3493.34	2.65%	3241.86	3.64%	930.90	5.98%
	TOTAL	745857.00		131690.77		89045.60		15579.26	
Avg. N Load Rate by Land Cover (kg/yr/ha)	Agricultural Land	28		36		24		32.94	
	Forest	2		9		6		NA	
	Grassland	2		6		4		5.6294	
	NO ESV	13		23		14		23.51	
	Open Water	21		29		19		29.07	
	Pasture	22		27		20		55.58	
	Shrub/Scrub	4		11		9		13.15	
	Urban Green Space	7		12		4		7.00	
	Wetland	11		17		13		19.83	
TOTAL N Load by Source (kg/yr)	Cafo	1362733	39.79%	1217481	61.69%	286230	42.57%	144619	66.43%
	Farm	1736113	50.69%	675411	34.22%	342491	50.94%	66752	30.66%
	Non-Farm	10580	0.31%	2322	0.12%	1052	0.16%	108	0.05%
	Non-Sewer	21408	0.63%	7385	0.37%	3189	0.47%	389	0.18%
	Atmospheric	194952	5.69%	47492	2.41%	29030	4.32%	5464	2.51%
	Past	37547	1.10%	8065	0.41%	3022	0.45%	380	0.17%
	Range	61746	1.80%	15427	0.78%	7321	1.09%	0	0.00%
TOTAL N Load by Land Cover (kg/yr)	Agricultural Land	2522105	73.64%	1530344	77.54%	559935	83.28%	193917.03	89.07%
	Forest	15393	0.45%	94	0.00%	2	0.00%	NA	NA
	Grassland	47432	1.38%	7696	0.39%	4665	0.69%	923.26	4.83%
	NO ESV	191671	5.60%	127214	6.45%	35583	5.29%	10525.81	0.96%
	Open Water	37118	1.08%	24932	1.26%	6135	0.91%	2093.40	0.86%
	Pasture	367295	10.72%	217193	11.01%	27360	4.07%	1872.75	2.64%
	Shrub/Scrub	173727	5.07%	42621	2.16%	26638	3.96%	5739.05	0.00%
	Urban Green Space	7308	0.21%	3658	0.19%	487	0.07%	4.12	1.211%
	Wetland	63029	1.84%	19830	1.00%	11530	1.71%	2636.78	1.21%
	TOTAL N LOAD (kg/yr)	3425078.54		1973581.91		672335.84		217712.21	
ESV (LOW) by Land Cover ($/yr)	Agricultural Land	$34,877,709	6.29%	$13,683,561	14.27%	$7,745,876	9.68%	1544325.63	7.57%
	Forest	$68,952,866	12.44%	$23,117	0.02%	$4,005	0.01%	NA	NA
	Grassland	$16,437,997	2.97%	$1,848,247	1.93%	$1,386,028	1.73%	228695.65	1.12%
	NO ESV	$0	0.00%	$0	0.00%	$0	0.00%	0.00	0.00%
	Open Water	$7,026,743	1.27%	$2,997,244	3.13%	$1,836,401	2.30%	408263.95	2.00%
	Pasture	$1,173,610	0.21%	$490,206	0.51%	$92,553	0.12%	1641.41	0.01%
	Shrub/Scrub	$80,860,530	14.59%	$7,531,018	7.86%	$8,018,458	10.02%	1306982.85	6.41%
	Urban Green Space	$18,755,228	3.38%	$5,875,321	6.13%	$2,063,477	2.58%	13831.08	0.07%
	Wetland	$326,235,549	58.85%	$63,410,703	66.15%	$58,845,999	73.56%	16897681.45	82.83%
	TOTAL	$554,320,233		$95,859,418		$79,992,796		$20,401,422	
ESV (HIGH) by Land Cover ($/yr)	Agricultural Land	$255,364,257	6.39%	$100,186,979	18.93%	$56,713,012	12.15%	$11,307,095	12.18%
	Forest	$231,750,779	5.80%	$77,696	0.01%	$13,461	0.00%	NA	NA
	Grassland	$16,437,997	0.41%	$1,848,247	0.35%	$1,386,028	0.30%	$228,696	0.25%
	NO ESV	$0	0.00%	$0	0.00%	$0	0.00%	$0	0.00%
	Open Water	$58,205,894	1.46%	$24,827,616	4.69%	$15,211,796	3.26%	$3,381,847	3.64%
	Pasture	$29,133,039	0.73%	$12,168,614	2.30%	$2,297,478	0.49%	$40,746	0.04%
	Shrub/Scrub	$2,720,882,487	68.09%	$253,411,836	47.89%	$269,813,726	57.81%	$43,978,771	47.37%
	Urban Green Space	$30,450,309	0.76%	$9,538,958	1.80%	$3,350,186	0.72%	$22,456	0.02%
	Wetland	$653,987,979	16.37%	$127,116,243	24.02%	$117,965,611	25.27%	$33,873,931	36.49%
	TOTAL	$3,996,212,742		$529,176,188		$466,751,297		$92,833,541	

Table 7: Aggregated Land Cover Summary

Land Cover Summary		HUC 8		GWMA		HUC10		HUC12	
		LYRB	% of total	GWMA	% of total	Spring Creek - Yakima River	% of total	Horseshoe Lake - Yakima River	% of total
Land Cover (Hectares)	Area of Complex Channels	1.15	0.0002%						
	Barren Land	178.44	0.02%						
	Canal/Ditch	244.53	0.03%	138.48	0.11%	24.07	0.02%		
	Canal/Ditch; Aqueduct	0.26	0.00003%	0.26	0.0002%	0.16	0.00%		
	Cultivated Crops	150224.87	20.14%	58937.68	44.75%	37292.52	24.38%	6651.70	42.70%
	Deciduous Forest	205.31	0.03%	4.96	0.004%				
	Developed, High Intensity	869.40	0.12%	110.99	0.08%	28.82	0.02%		
	Developed, Low Intensity	19084.90	2.56%	4835.69	3.67%	2014.96	1.32%	239.89	1.54%
	Developed, Medium Intensity	6368.72	0.85%	987.79	0.75%	371.53	0.24%	9.00	0.06%
	Developed, Open Space	23461.82	3.15%	9627.43	7.31%	4771.70	3.12%	526.30	3.38%
	Emergent Herbaceous Wetlands	4159.33	0.56%	993.62	0.75%	715.96	0.47%	13.98	0.09%
	Evergreen Forest	67771.29	9.09%	16.70	0.01%	3.96	0.00%		
	Freshwater Emergent Wetland	4564.84	0.61%	869.74	0.66%	1105.58	0.72%	300.90	1.93%
	Freshwater Forested/Shrub Wetland	3478.32	0.47%	411.40	0.31%	483.89	0.32%	70.00	0.45%
	Freshwater Pond	713.35	0.10%	235.90	0.18%	242.61	0.16%	54.31	0.35%
	Grassland/Herbaceous	45711.89	6.13%	5139.73	3.90%	3897.30	2.55%	635.97	4.08%
	Lake	769.05	0.10%	405.42	0.31%	447.38	0.29%	82.58	0.53%
	Lake/Pond; Intermittent	78.47	0.01%	33.88	0.03%	34.53	0.02%	1.45	0.01%
	Lake/Pond; Perennial	137.04	0.02%	40.35	0.03%	29.59	0.02%	10.20	0.07%
	Lake/Pond; Perennial avg level	0.90	0.0001%						
	Mixed Forest	219.28	0.03%	1.21	0.001%				
	Open Water	434.38	0.06%	248.06	0.19%	155.81	0.10%	22.87	0.15%
	Other (Water)	9.60	0.001%	1.13	0.001%	0.42	0.00%		
	Pasture/Hay	24093.82	3.23%	10063.78	7.64%	1940.40	1.27%	33.70	0.22%
	Reservoir; filtration pond	6.29	0.001%	6.29	0.005%	6.29	0.00%		
	Reservoir; nonearthen, covered	0.20	0.00003%						
	Reservoir; settling pond	6.10	0.001%	5.91	0.004%				
	Reservoir; sewage treament pond	5.63	0.001%	5.40	0.004%	5.12	0.00%		
	Reservoir; storage	0.98	0.0001%	0.98	0.001%				
	Riverine	2033.36	0.27%	770.85	0.59%	603.68	0.39%	101.99	0.65%
	Shrub/Scrub	383297.93	51.39%	35698.80	27.11%	97418.78	63.68%	6195.41	39.77%
	Stream/River	1.41	0.0002%	0.15	0.0001%				
	Stream/River Perennial	322.60	0.04%	86.14	0.07%	85.51	0.06%	12.20	0.08%
	Swamp/Marsh; Intermittent	307.47	0.04%	3.31	0.003%	0.41	0.00%	0.00	0.00%
	Swamp/Marsh; Perennial	347.04	0.05%	27.25	0.02%	48.39	0.03%	4.24	0.03%
	Water areas	1631.49	0.22%	793.50	0.60%	229.02	0.15%	70.78	0.45%
	Woody Wetlands	5115.52	0.69%	1188.03	0.90%	1030.15	0.67%	541.78	3.48%

Table 8: Average N Loading Rates for Aggregated Land Cover Dataset

		Avg. N Load Rates			
		HUC 8	GWMA	HUC 10	HUC 12
		LYRB	GWMA	Spring Creek - Yakima River	Horseshoe Lake - Yakima River
Avg. N Load Rate by Land Cover (kg/yr/ha)	Area of Complex Channels	3			
	Barren Land	1			
	Canal/Ditch	19	23	12	
	Canal/Ditch; Aqueduct	6	6	8	
	Cultivated Crops	28	36	24	33
	Deciduous Forest	2	3		
	Developed, High Intensity	7	32	4	
	Developed, Low Intensity	13	24	15	25
	Developed, Medium Intensity	6	17	9	22
	Developed, Open Space	13	21	13	22
	Emergent Herbaceous Wetlands	8	11	10	13
	Evergreen Forest	2	11	6	
	Freshwater Emergent Wetland	14	23	15	23
	Freshwater Forested/Shrub Wetland	11	22	12	15
	Freshwater Pond	16	27	15	28
	Grassland/Herbaceous	2	6	4	6
	Lake	11	11	11	7
	Lake/Pond; Intermittent	14	19	10	21
	Lake/Pond; Perennial	8	18	7	8
	Lake/Pond; Perennial avg level	2			
	Mixed Forest	1	4		
	Open Water	9	14	6	9
	Other (Water)	7	8	4	
	Pasture/Hay	22	27	20	56
	Reservoir; filtration pond	5	5	5	
	Reservoir; nonearthen, covered	3			
	Reservoir; settling pond	12	12		
	Reservoir; sewage treament pond	3	3	4	
	Reservoir; storage	7	7		
	Riverine	19	24	8	11
	Shrub/Scrub	4	11	9	13
	Stream/River	19	5		
	Stream/River Perennial	5	4	5	3
	Swamp/Marsh; Intermittent	9	22	0	0
	Swamp/Marsh; Perennial	19	36	11	22
	Water areas	22	29	22	30
	Woody Wetlands	7	11	11	13

Table 9: Total N Loading for Aggregated Land Cover Dataset

	Total N Loading Summary								
		HUC 8		GWMA		HUC 10		HUC 12	
		LYRB	% of total	GWMA	% of total	Spring Creek - Yakima River	% of total	Horseshoe Lake - Yakima River	% of total
TOTAL N Load by Land Cover (kg/yr)	Area of Complex Channels	2	0.00002%						
	Barren Land	105	0.001%						
	Canal/Ditch	5592	0.07%	2060	0.10%	200	0.02%		
	Canal/Ditch; Aqueduct	1	0.00002%	1	0.00003%	1	0.00%		
	Cultivated Crops	5576843	74.55%	1531124	77.55%	736279	83.63%	222342.54	89.95%
	Deciduous Forest	341	0.005%	5	0.0002%				
	Developed, High Intensity	4871	0.07%	1348	0.07%	47	0.01%		
	Developed, Low Intensity	142505	1.90%	43859	2.22%	12884	1.46%	4785.47	1.94%
	Developed, Medium Intensity	23501	0.31%	4255	0.22%	791	0.09%	171.58	0.07%
	Developed, Open Space	233162	3.12%	79263	4.01%	25100	2.85%	6108.51	2.47%
	Emergent Herbaceous Wetlands	21364	0.29%	3361	0.17%	2020	0.23%	52.96	0.02%
	Evergreen Forest	30279	0.40%	86	0.00%	2	0.00%		
	Freshwater Emergent Wetland	62615	0.84%	9763	0.49%	7350	0.83%	1483.11	0.60%
	Freshwater Forested/Shrub Wetland	21404	0.29%	3295	0.17%	1104	0.13%	224.70	0.09%
	Freshwater Pond	9424	0.13%	2554	0.13%	1040	0.12%	247.65	0.10%
	Grassland/Herbaceous	95848	1.28%	7696	0.39%	4910	0.56%	931.77	0.38%
	Lake	2146	0.03%	468	0.02%	633	0.07%	58.52	0.02%
	Lake/Pond; Intermittent	994	0.01%	337	0.02%	161	0.02%	3.57	0.00%
	Lake/Pond; Perennial	1341	0.02%	263	0.01%	140	0.02%	26.82	0.01%
	Lake/Pond; Perennial avg level	1	0.00002%						
	Mixed Forest	229	0.003%	3	0.0002%				
	Open Water	3034	0.04%	1103	0.06%	442	0.05%	71.73	0.03%
	Other (Water)	27	0.0004%	8	0.0004%	0	0.00%		
	Pasture/Hay	800264	10.70%	217195	11.00%	33672	3.82%	1939.75	0.78%
	Reservoir; filtration pond	16	0.0002%	8	0.0004%	8	0.00%		
	Reservoir; nonearthen, covered	0	0.000005%						
	Reservoir; settling pond	171	0.002%	85	0.004%				
	Reservoir; sewage treament pond	36	0.0005%	17	0.001%	17	0.00%		
	Reservoir; storage	6	0.0001%	2	0.0001%				
	Riverine	13433	0.18%	4321	0.22%	504	0.06%	43.84	0.02%
	Shrub/Scrub	357881	4.78%	42638	2.16%	46771	5.31%	5928.84	2.40%
	Stream/River	16	0.0002%	0	0.00001%				
	Stream/River Perennial	1328	0.02%	106	0.005%	109	0.01%	8.55	0.00%
	Swamp/Marsh; Intermittent	2912	0.04%	81	0.004%	0	0.00%	0.00	0.00%
	Swamp/Marsh; Perennial	7854	0.10%	717	0.04%	543	0.06%	22.13	0.01%
	Water areas	44655	0.60%	15797	0.80%	3843	0.44%	1753.22	0.71%
	Woody Wetlands	16810	0.22%	2623	0.13%	1828	0.21%	988.01	0.40%

Figure 17: Estimated Nitrogen Load and ESV (Low Median)

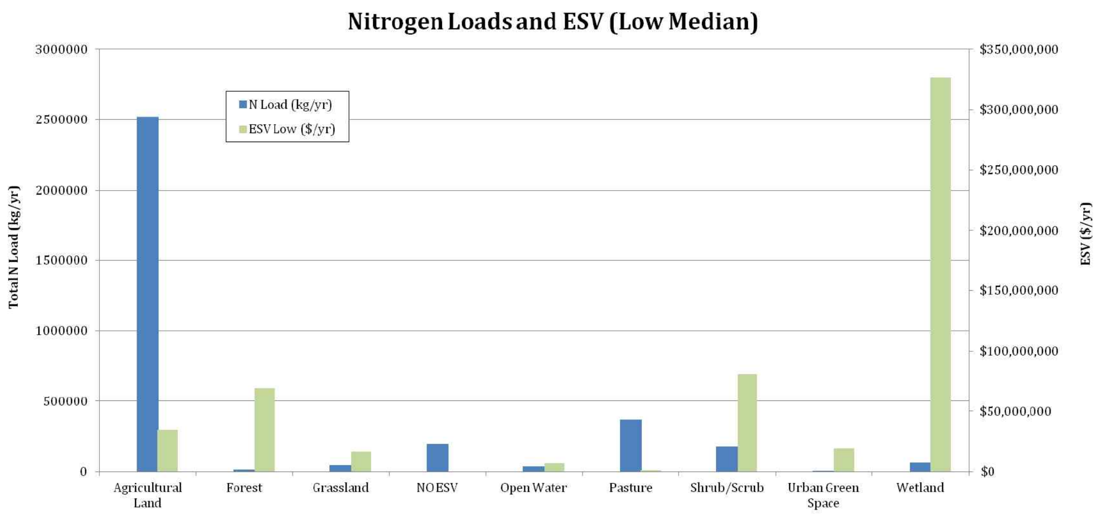


Figure 18: Estimated Nitrogen Load and ESV (High Median)

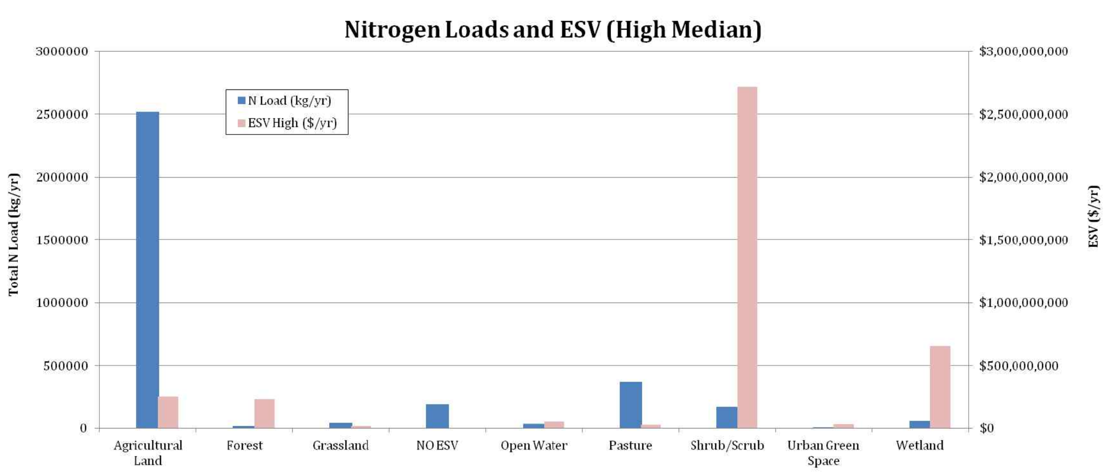


Figure 19: Total N Load and Total ESV (Low Median) by HUC 12

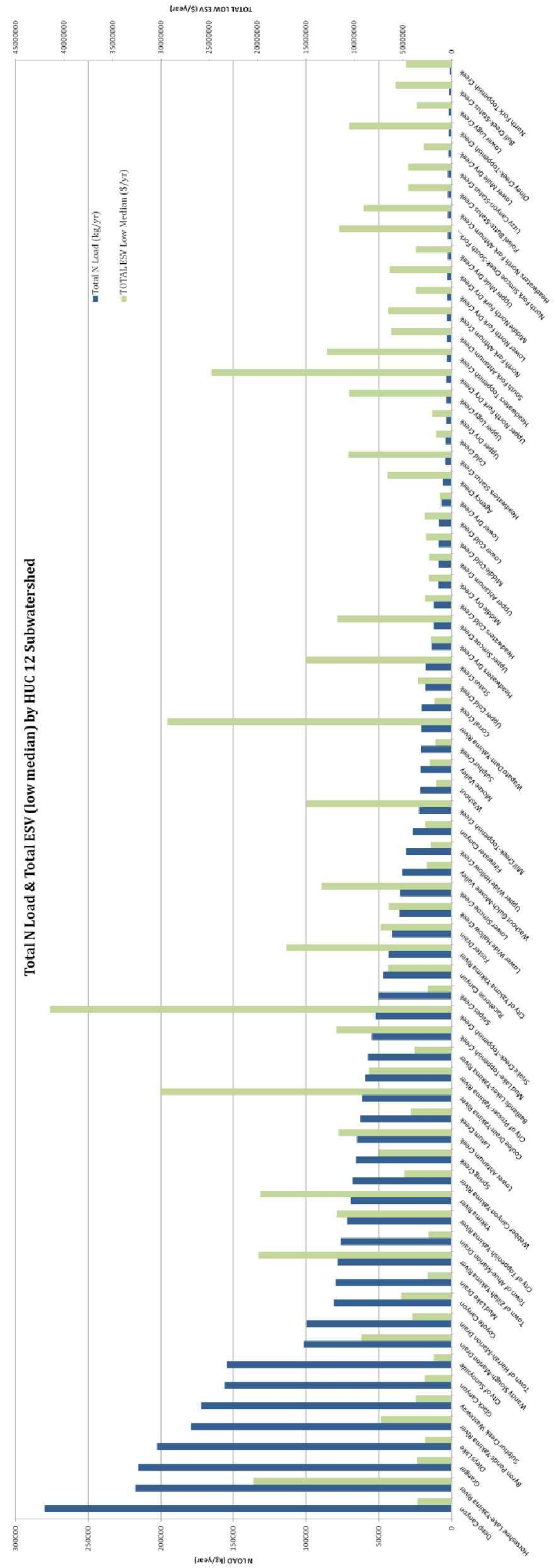


Figure 20: Total N Load and Total ESV (High Median) by HUC 12

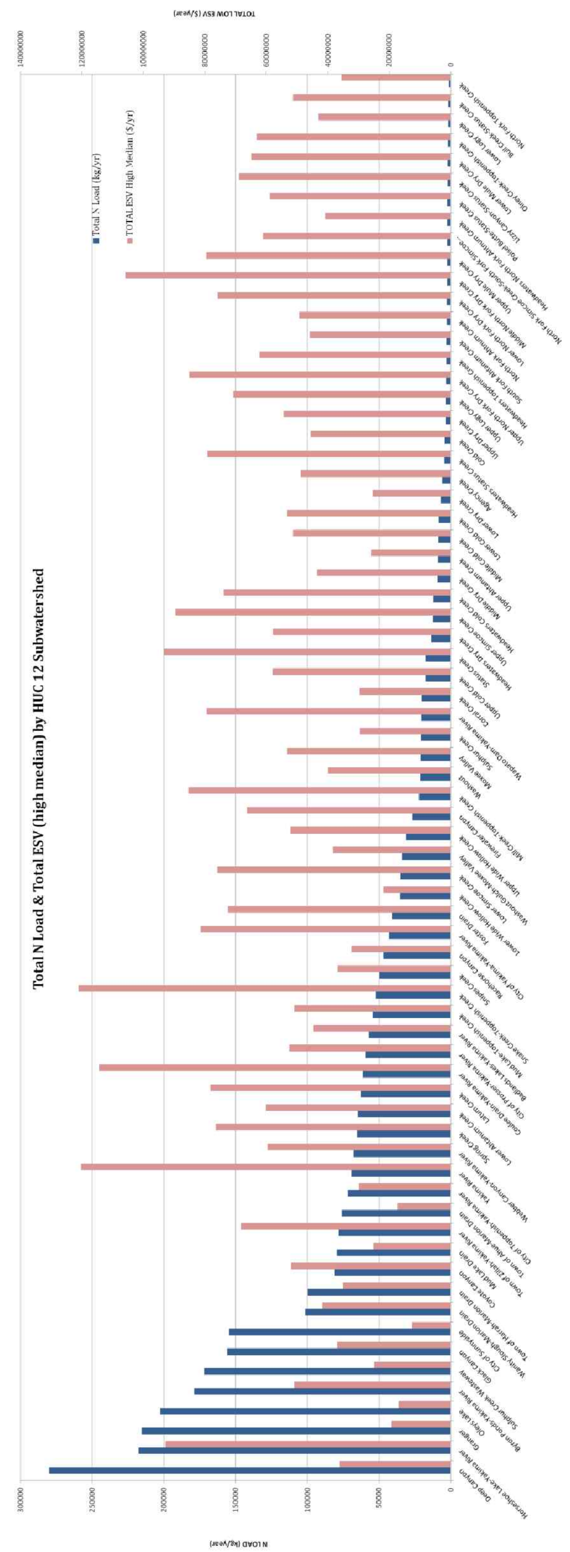


Figure 21: Total N Load and Total ESV (Low Median) by HUC 10

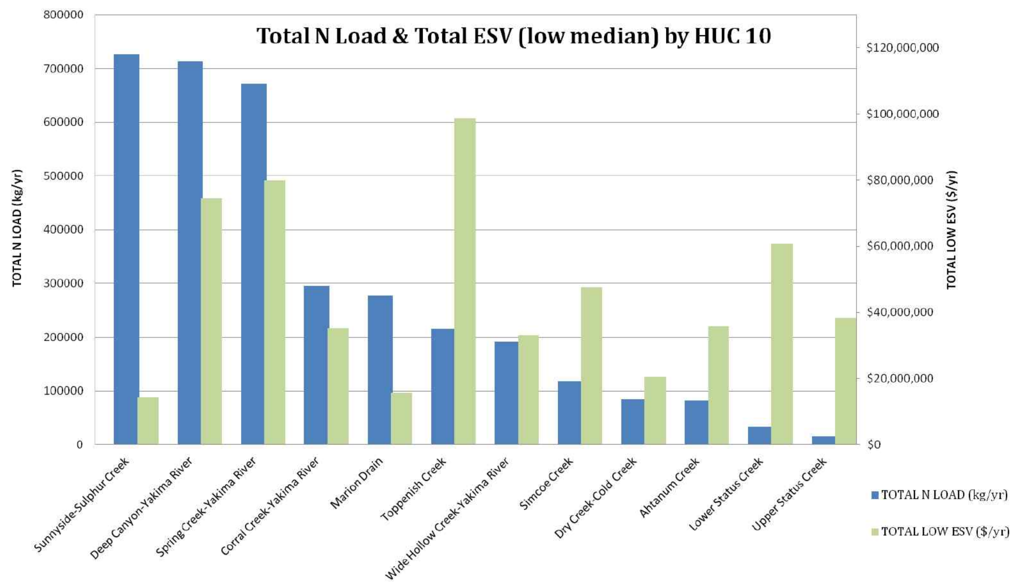


Figure 22: Total N Load and Total ESV (High Median) by HUC 10

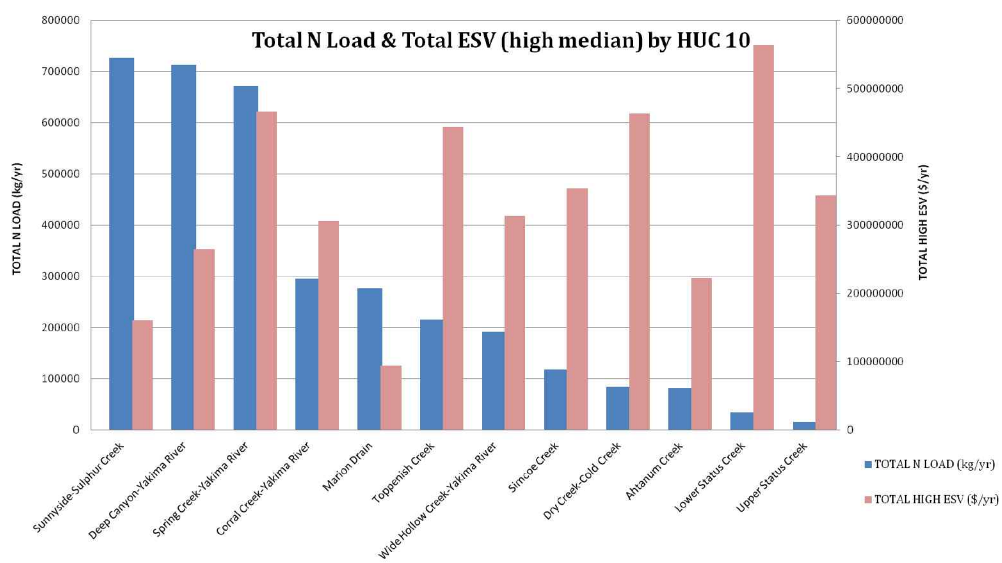


Figure 23: Area and Total N Load by Land Cover (GWMA)

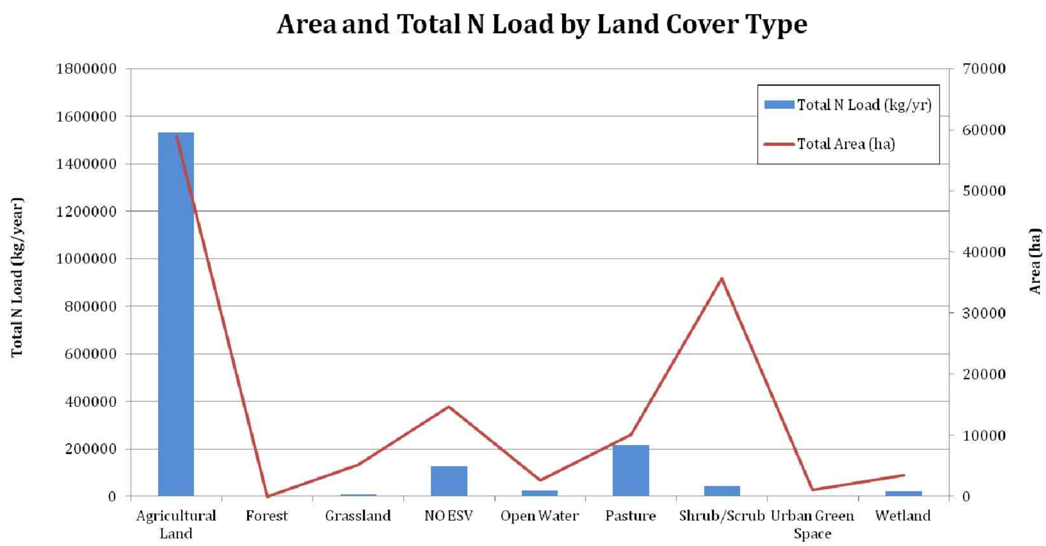


Figure 24: Total N Load by Source – GWMA

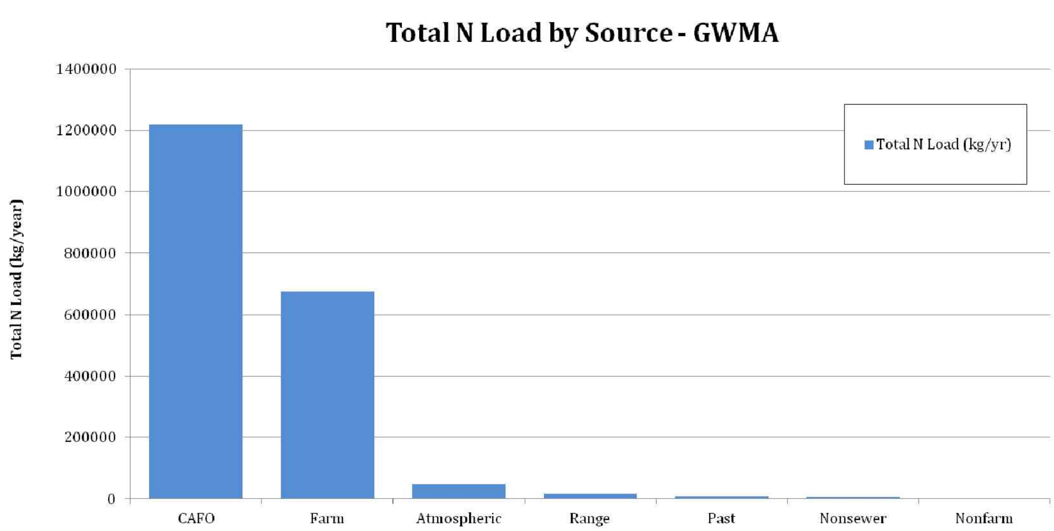


Figure 25: Area and Total N Load by Land Cover (Spring Creek)

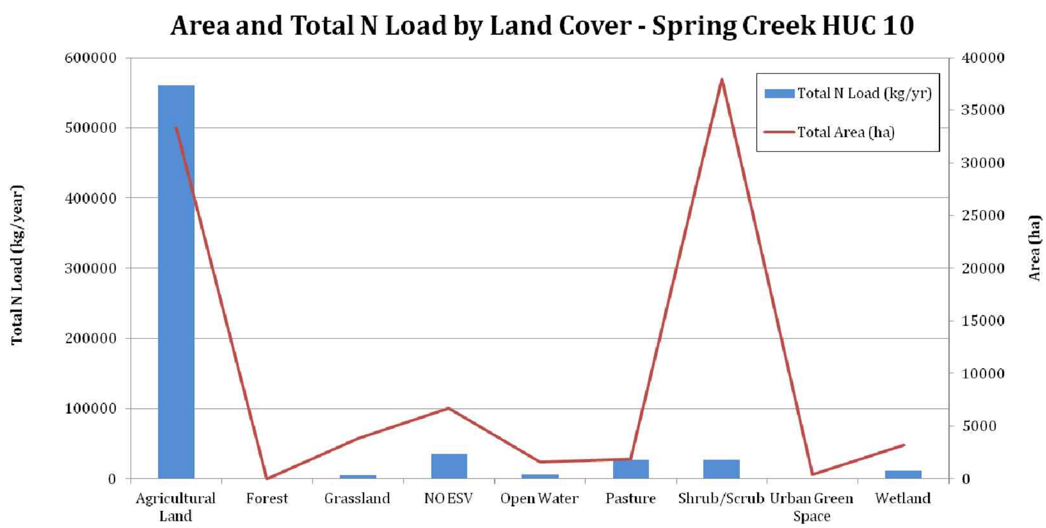


Figure 26: Total N Load by Source (Spring Creek)

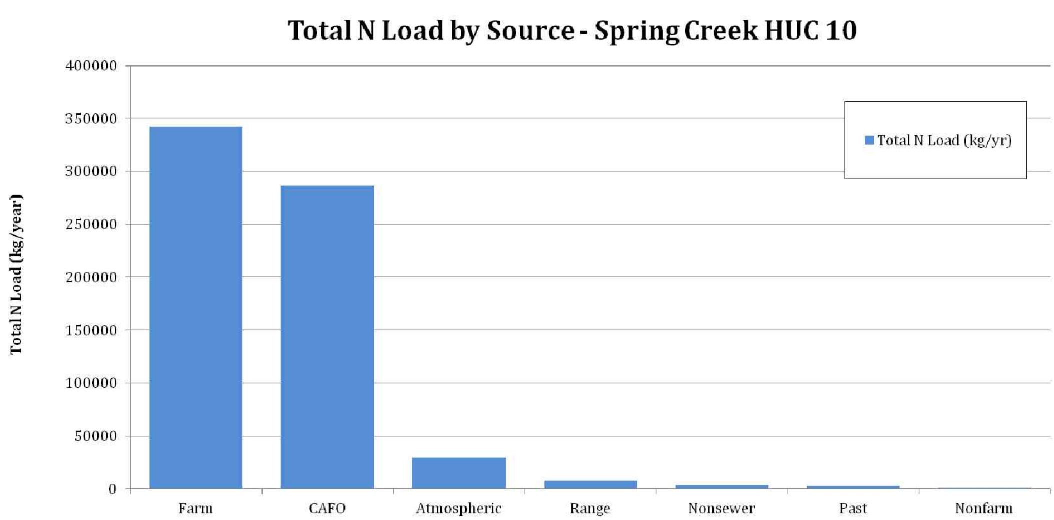


Figure 27: Area and Total N Load by Land Cover (Horseshoe Lake)

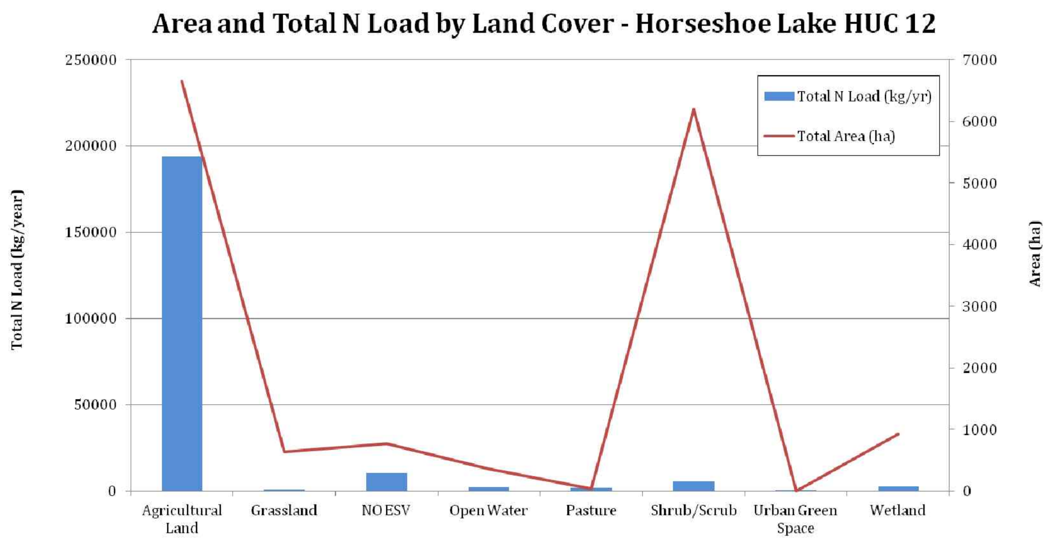


Figure 28: Total N Load by Source (Horseshoe Lake)

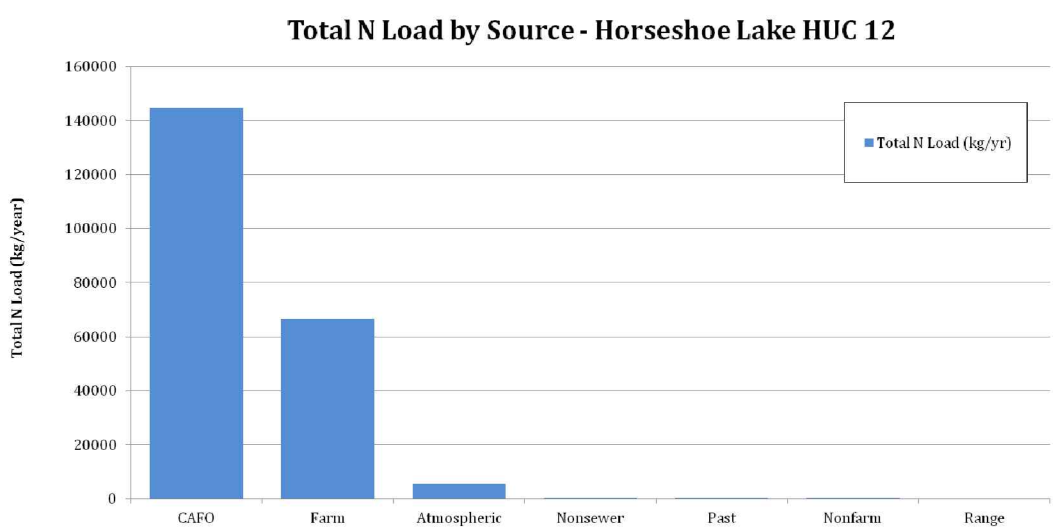


Figure 29: Average N Loading Rates (GWMA)

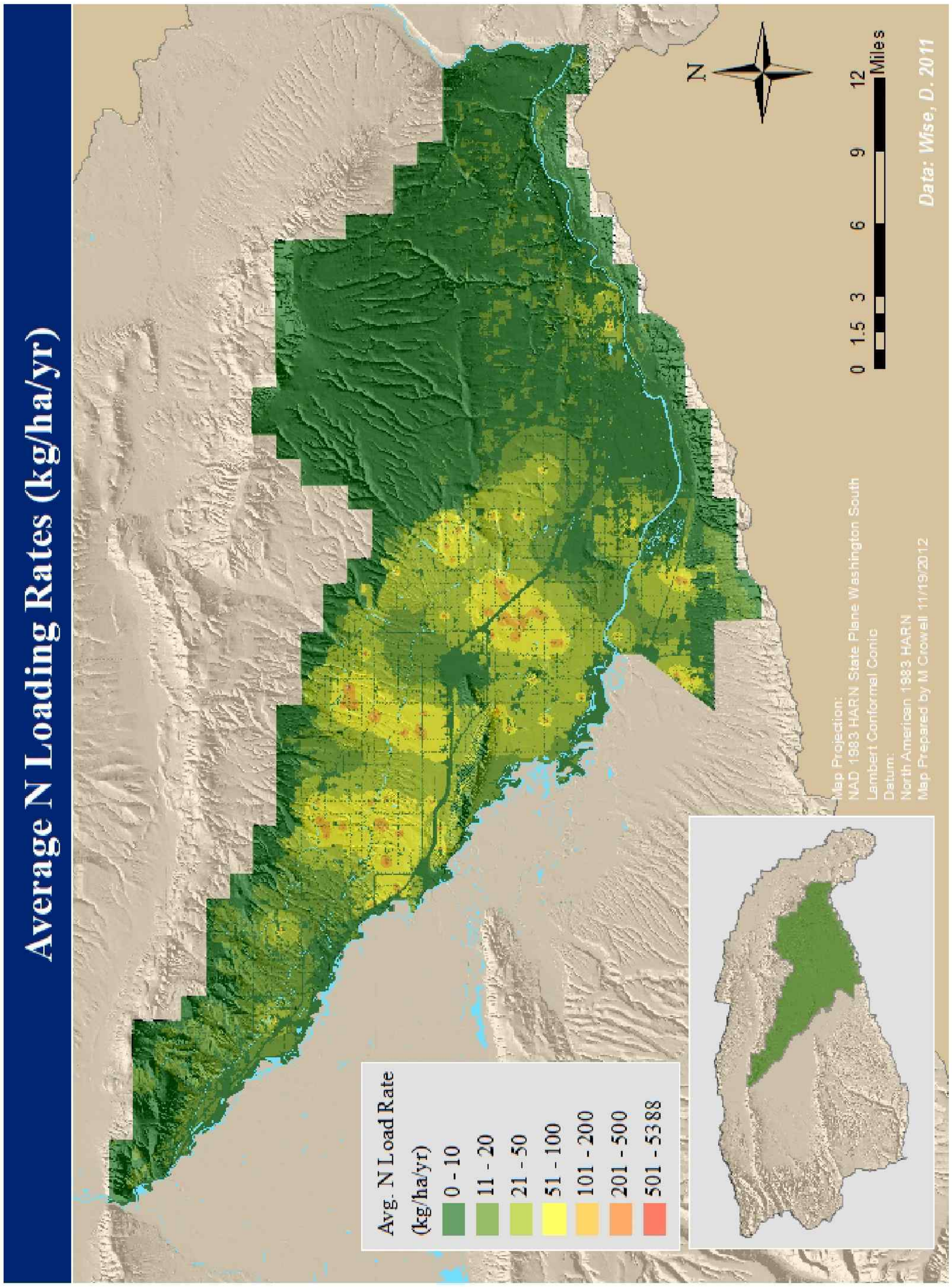


Figure 30: Average N Loading Rates (Spring Creek)

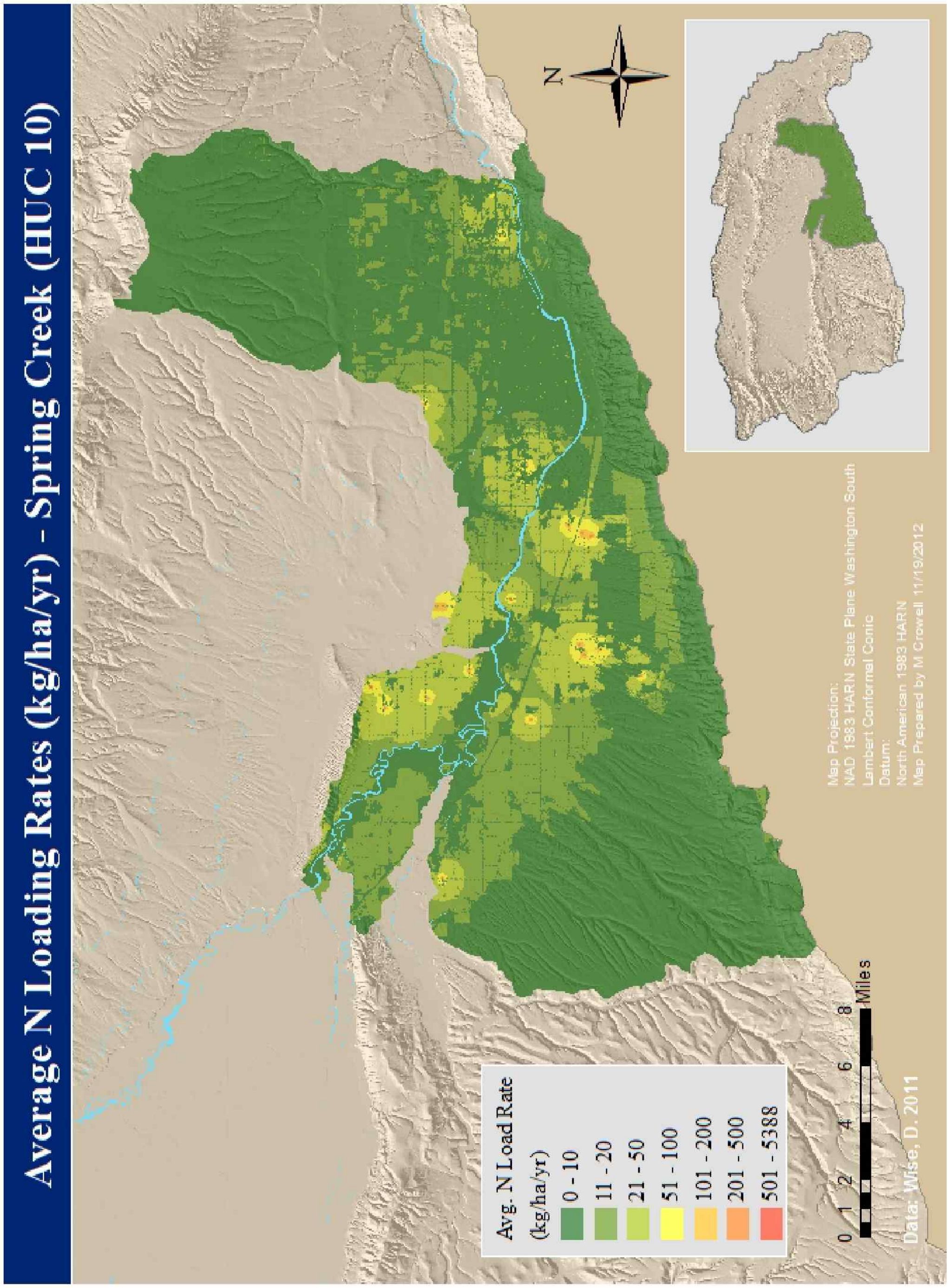


Figure 31: Average N Loading Rates (Horseshoe Lake)

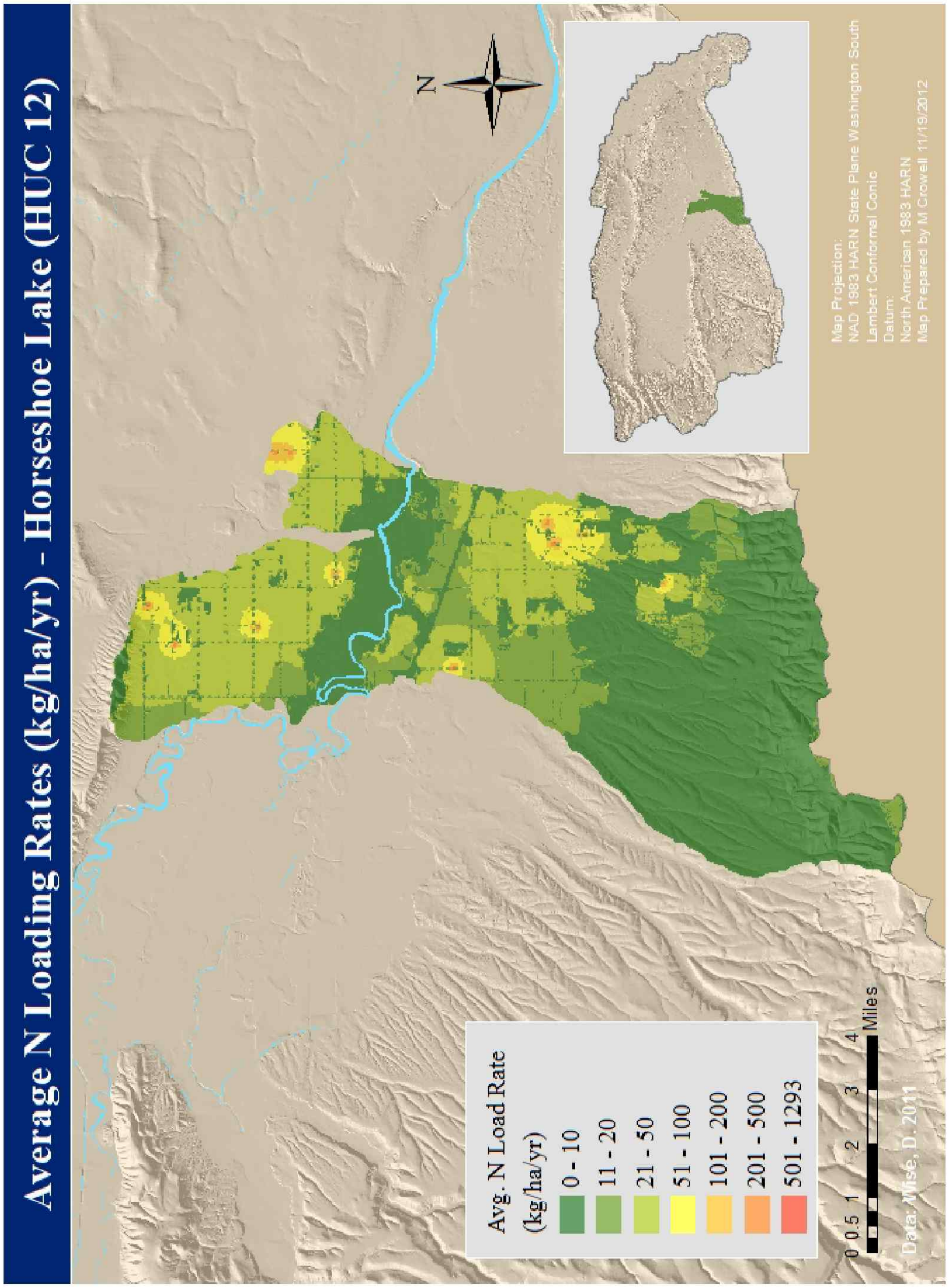


Figure 32: Ecosystem Service Values (Low Median, GWMA)

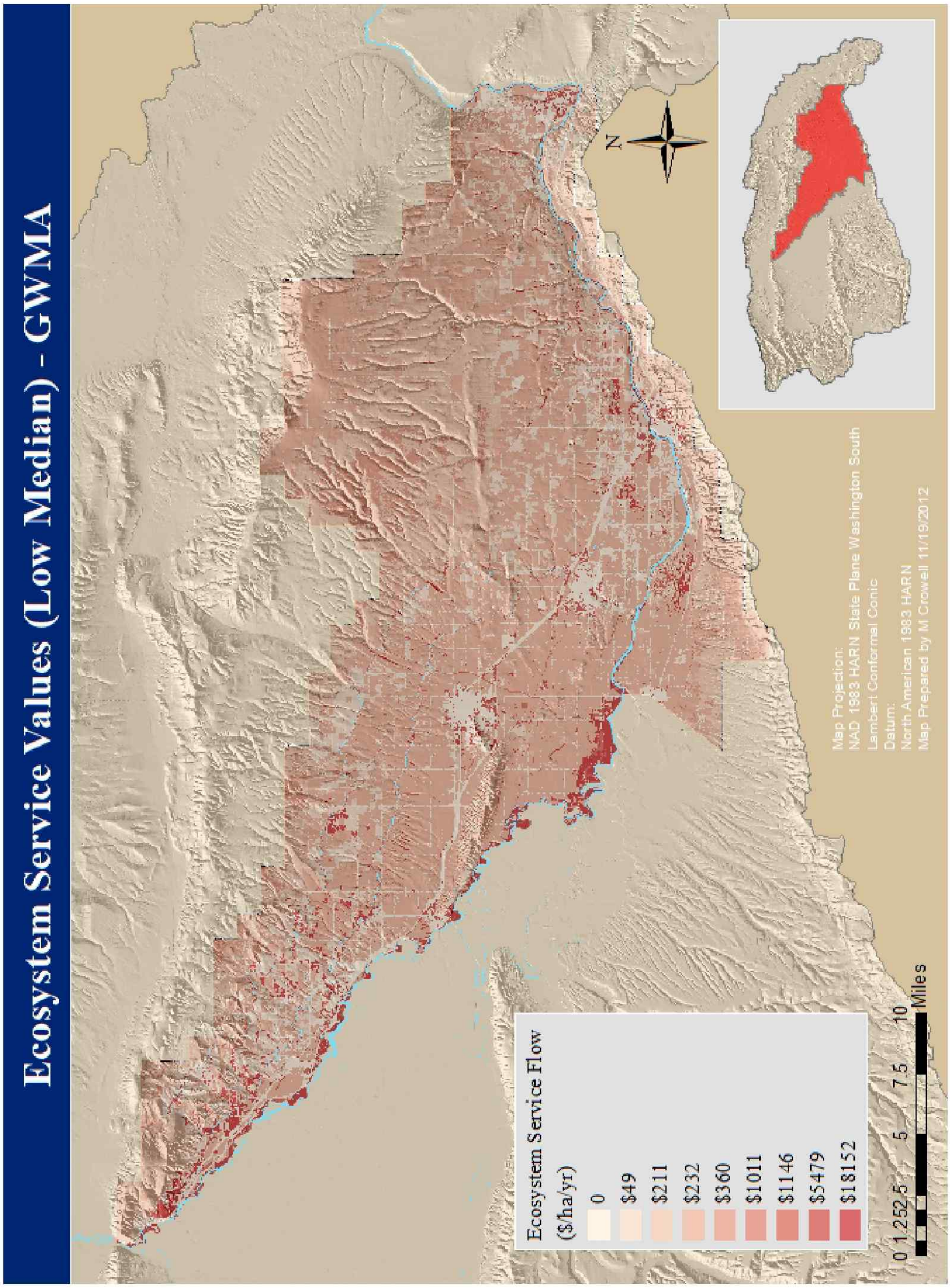


Figure 33: Ecosystem Service Values (Low Median, Spring Creek)

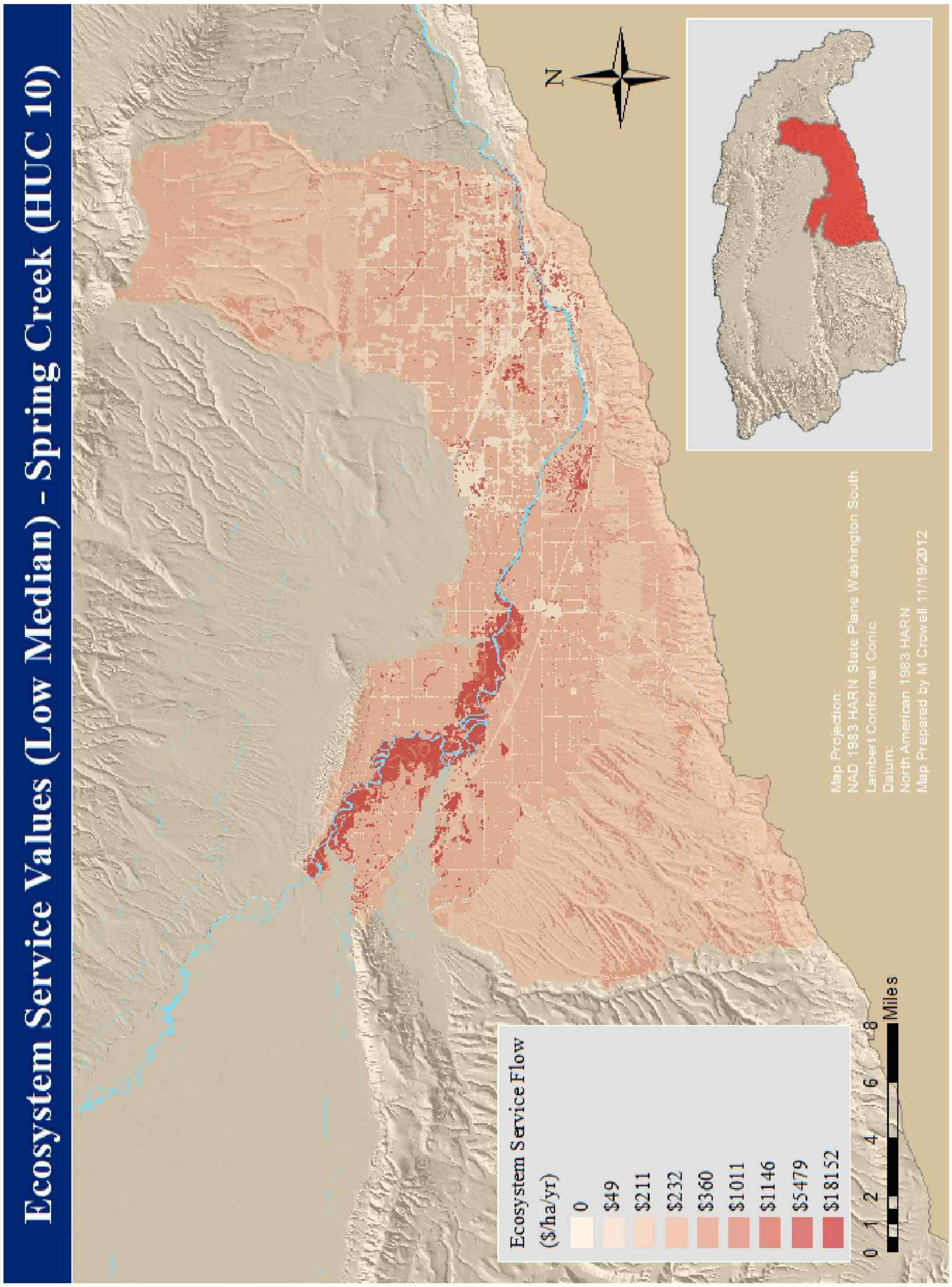
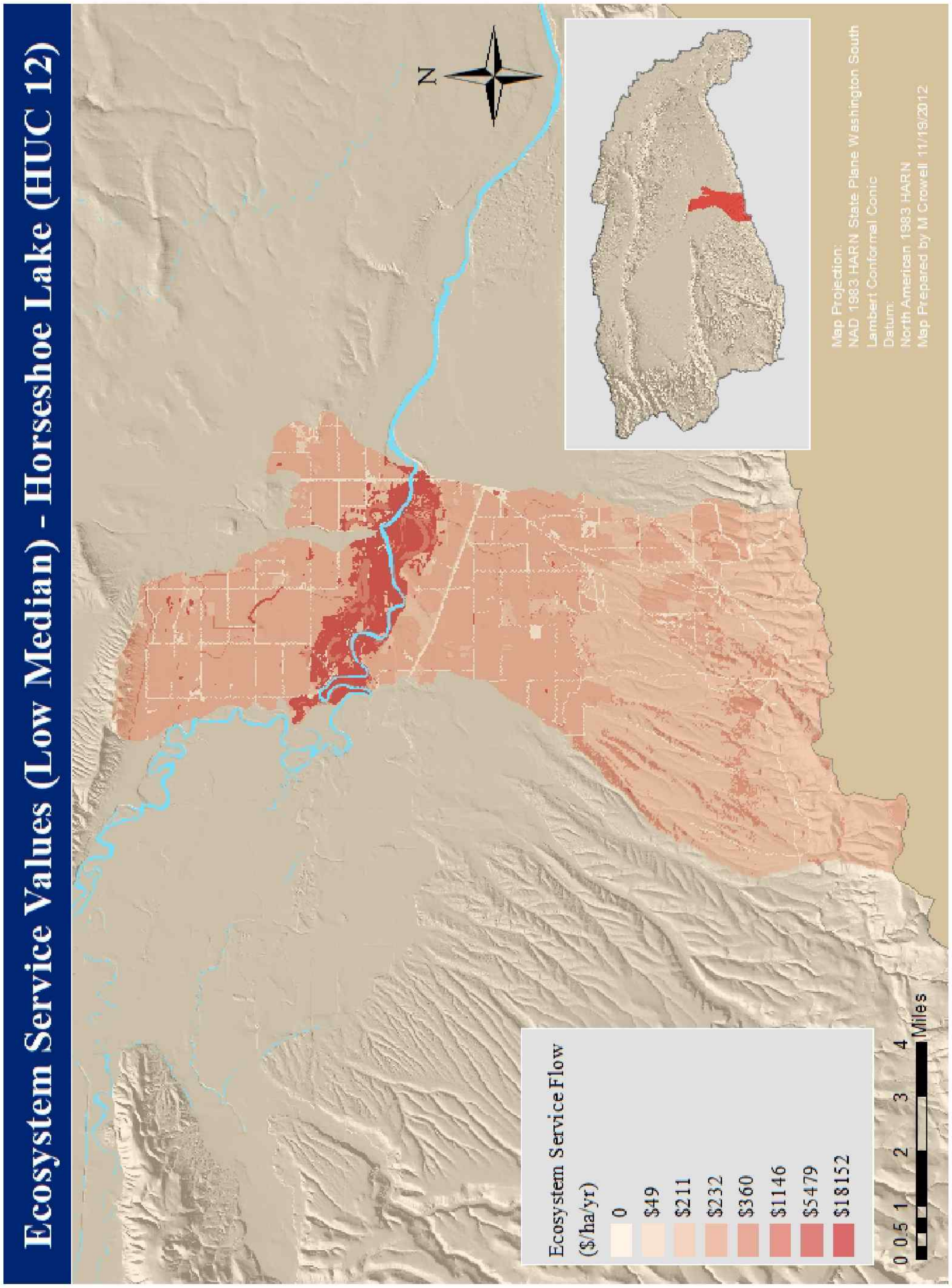


Figure 34: Ecosystem Service Values (Low Median, Horseshoe Lake) (65)



Results Section 4.5: Groundwater Ecosystem Service Valuation

After performing the ecosystem service and N loading analysis the final component of groundwater ecosystem services was evaluated using the Crutchfield et al. 1997 regression model. Each of the variables for this function were updated as shown below (with function coefficients):

- ***Age (-0.014442):*** 2010 Census
- ***Sex (0.409802):*** 2010 Census
- ***Personal Income (0.000000):*** 2010 Census (Mean Non-Family)
- ***Household Income (NA):*** 2010 Census (Mean)
- ***Extra Income (0.000004):*** Household Income – Personal Income
- ***Education Level (0.067758):*** 2010 Census
- ***Rural (0.251104):*** For census tracts located within city boundaries, 10% of total population was assumed to live in rural area. For census tracts outside of city boundaries, 90% of total population was assumed to live in a rural area. (Crutuchfield et al. 1997 mean value = 0.22)
- ***Connection to Municipal System (-0.132924):*** For census tracts located within city boundaries, 90% was assumed connected. For census tracts outside of city boundaries, 0% assumed connected. (Crutchfield et al. 1997 mean value = 0.74)
- ***Aware of Nitrate Issues (0.077777):*** Assumed a 10% increase from Crutchfield et al. 1997 mean values due to increased public attention on nitrate in the LYRB. (Crutchfield et al. 1997 mean value = 0.23)
- ***Uses at-home treatment system (-0.305657):*** Crutchfield et al. 1997 mean values
- ***Use of Bottled Water (0.089862):*** Crutchfield et al. 1997 mean values
- ***Years at Zip Code (0.097674):*** Crutchfield et al. 1997 mean values

While assumptions were made regarding rural population statistics and connection to municipal systems, these assumptions were made based upon population distribution and extent of municipal systems in the LYRB. In addition to the potential age issues with Crutchfield et al. 1997 values, these values were also generated in a much larger area that included several urban centers so these values are not likely to accurately represent the LYRB. Additional assumptions regarding awareness of nitrate issues were likely conservative and were made due to the age of the Crutchfield et al. (1997) study and the recent campaigns of local government agencies and community organizations to increase awareness of these issues in the study area. Unfortunately data for at-home system treatment, use of bottled water and years at zip code were not located and there was no way to conservatively estimate changes to the 1997 mean data based upon study site characteristics. Detailed function variable values for each census tract used are shown in Appendix 3 along with function coefficients and variable definition data shown in Appendix 4-5.

With the updated function values, willingness to pay (WTP) for "safe" drinking water (nitrate-N level at or below the US EPA MCL of 10 mg/L) was estimated for 2010 census tracts within the study area. Several census tracts were partially located in the LYRB, but contained significant areas outside of the study site, so these areas were not evaluated as part of the WTP estimates. In total, household and total WTP were calculated for 43 census tracts (41 in Yakima County, 2 in Benton County). The estimated total annual WTP for these areas is $57,526,003 ($4,793,834 monthly). While the annual household WTP by census tract varies between $613.12 - $910.72, for the entire LYRB the average annual household WTP is $754. Figures 35 and 36 illustrate total and household WTP for "safe" drinking water by census tract.

By comparison, the Crutchfield 1997 study estimated average annual household WTP at $661.92. To address inflation concerns between these figures, we consider the regression function coefficients for personal income (0.000000) and extra income (0.000004). With the lack of strength of these coefficients in the valuation function differences in income due to inflation will not have large effects of WTP estimates. Additionally, because income levels and purchasing power are unlikely to have increased proportionally with inflation in the LYRB, adjusting the 1997 WTP values for inflation would not accurately represent the changes in income and WTP. Therefore, while a comparison between the 1997 WTP estimates and the updated valuation function does span across a number of years when inflation has occurred, adjusting values to account for inflation in these estimates is not necessary for the sake of comparison.

To frame these results within the ecosystem service framework, the WTP estimates indicate that if nitrate levels in groundwater throughout the entire LYRB exceeded the MCL, the benefit loss to the census tracts used would be approximately $57,526,003 annually, equivalent to $754 per household. Given that current studies estimate 34% of residents to be on private wells, with 12% of wells exceeding the MCL (PGG 2011), using the 2010 census average household size for Yakima County of 3.05, (Census 2010) current benefit loss due to nitrate levels in excess of the MCL is $720,161 per year.

Figure 35: Annual WTP for “Safer” Drinking Water

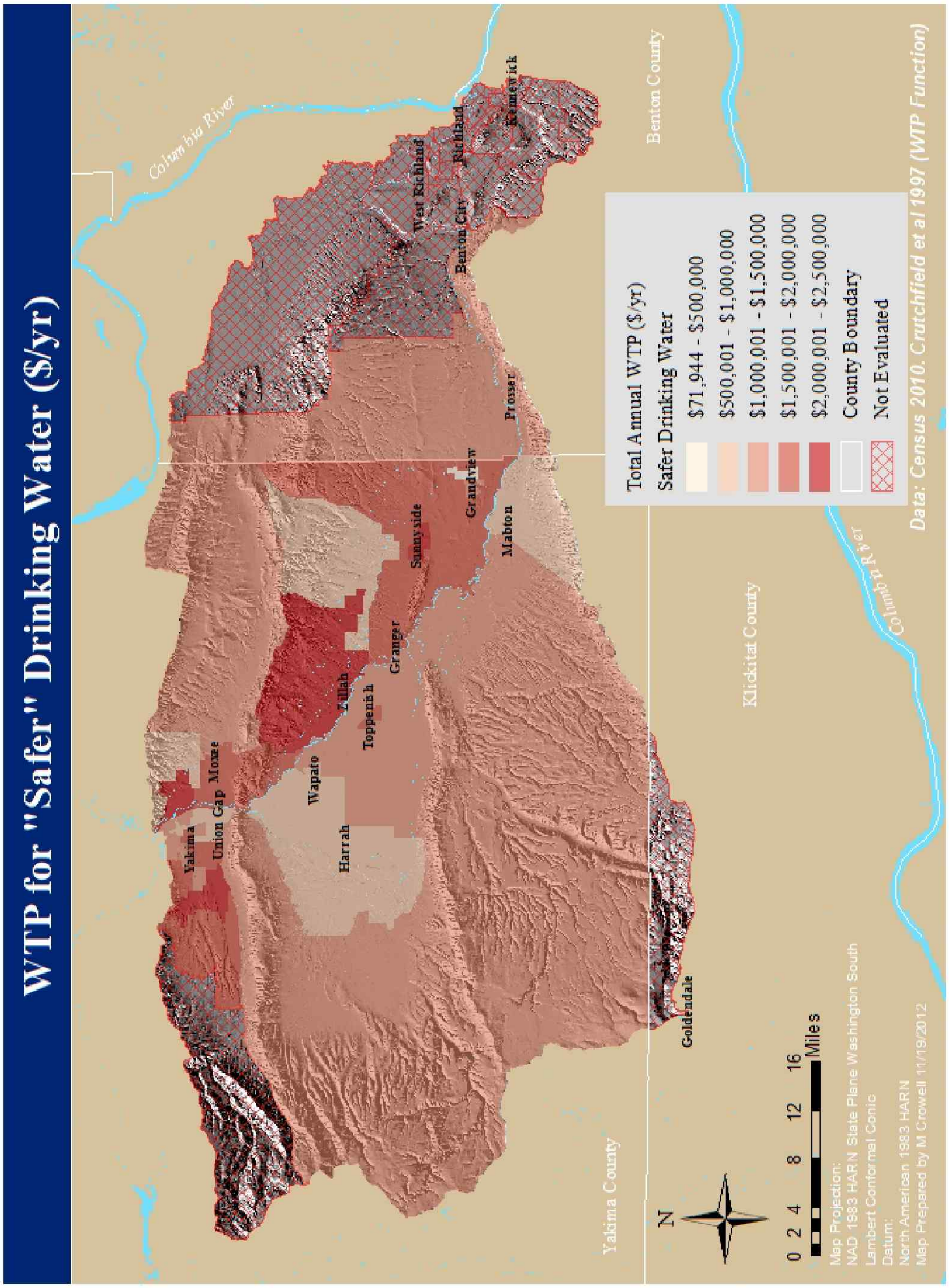
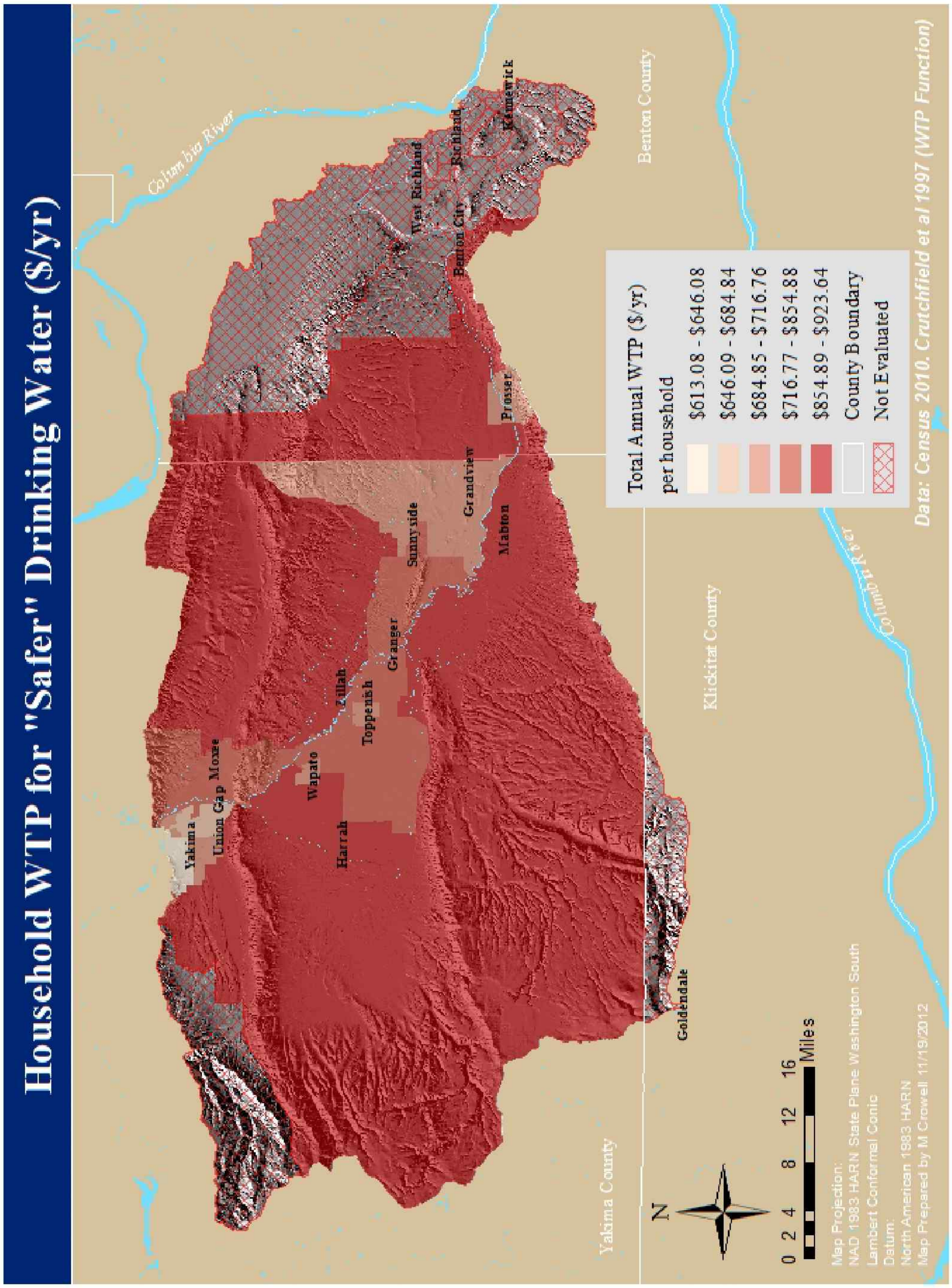


Figure 36: Household WTP for "Safer" Drinking Water



Discussion Section 5.1: Land Cover Analysis

Because the land cover analysis served as the basis for subsequent ecosystem service valuation and analysis of nitrogen loading data, the results of this analysis and potential errors must be considered. While most of the data processing and dataset ranking process ensured the highest accuracy with the available datasets, several components of the land cover classification and simplification process required analytical decision-making that may have affected subsequent valuation. Fortunately, these areas tended to represent small areas in the LYRB which likely did not generate significant affects in valuation accuracy, particularly when compared to potential errors in the ecosystem service valuation methodology as discussion in the following section.

The results of the land cover analysis clearly illustrate the importance of shrub/scrub (51.4%) and agricultural lands/cultivated crop (20.1%) land covers in the LYRB, with these accounting for 71.5% of total land area in all classifications with no simplification performed from the original NLCD 2006 data. Pasture (3.2%) and Grassland (6.1%) were the only other land cover types which did not require additional simplification and therefore are also only limited by the accuracy of the original NLCD 2006 dataset. Land cover types that did require simplification or additional modification, are potential sources of error due to over generalization or incorrect categorization. The first potential source of error, over generalization, may have occurred during the creation of the simplified land cover types of Forest (9.14%), Open Water (0.08%) and Wetlands (2.4%). In the each of these land covers, multiple land cover types were combined to create a single category for the purpose of valuation. Clearly in each of these processes the land cover types combined are compatible and accurately represented by the simplified typology. Therefore any errors associated with this data are due to valuation methodology discussed in the following section.

A more problematic situation arose with determining the land cover types to be included in the no ecosystem service value (NO ESV) classification. While “Developed” and “Barren” lands clearly represented land cover types associated with little to no ecosystem service value, other classifications such as reservoirs, and canal/ditch may in fact generate ecosystem services that were lost in valuation after designated as NO ESV. While “Developed, Low Intensity” (NLCD 22) may in fact generate ecosystem service values, for the sake of conservative estimates, all “Developed” land covers, with the exception of “Developed, Open Space” were designated as NO ESV. The justification for the NO ESV designation for the canal and reservoir land cover types, was justified by the very small areas where these land covers are present (0.00003-0.001%), combined with their unknown ability to provide ecosystem services, meant that by excluding them from valuation value estimates would remain conservative and the effects of not valuing these land covers would likely be small due to their size.

In addition to the NO ESV land cover classifications, a more discretionary choice was made to alter the land covers designated as “Urban Green Space” in the final simplified land cover. Originally, it was anticipated that the “Developed, Open Space” (NLCD 21) would serve as the basis for this land cover type, however upon inspection a great deal of land with this designation did not fit with the concept of urban green space as envisioned in this analysis. Specifically, a significant percentage of this land cover type was outside of urban areas, and therefore would be inaccurately described by the “Urban Green Space” category. To adjust the error, only areas of NLCD 21 that were found within city boundaries, as defined by the Washington State Department of Ecology 2010, were designated as “Urban Green Space” with areas outside these boundaries designated as “NO ESV.” While the potential still exists for NLCD 21 areas within city boundaries to be inaccurately described as “Urban Green Space” at least by clipping the data to within these boundaries these areas are minimized and kept within “urban” areas.

In addition to classification decisions made during land cover data analysis, other limitations of the data may be due to the datasets used in the analysis and any bias resulting from this data. While there was an intentional focus on including water features and wetlands, use of datasets focused on these, such as the National Hydrography Dataset and the National Wetlands Inventory may have resulted in either a more accurate estimation of these features, or potentially an over estimation due to more generous classifications of wetland features. The over estimation may have specifically occurred due to the fact that ecosystem condition is not represented in the data, and therefore highly degraded areas and former wetlands are equally represented with the healthiest and most productive wetlands. Clearly the connection between land cover designation and the ecological productivity of each parcel is critical to accurate ecosystem service valuation and is discussed in the following section 5.2. In order to create the most accurate land cover data for valuation, connecting satellite-based data with physical inventories and assessments of ecosystems and ground truthing of data would improve land cover analysis accuracy, but was beyond the scope of this project.

Discussion Section 5.2: Interpreting Ecosystem Service Valuation

The goal of the benefit transfer performed in this research was to conduct a general, preliminary valuation of ecosystem services in the LYRB to incorporate with analysis of nitrogen loading in the study site. While the benefit transfer results were sufficient to perform this analysis, there are numerous limitations of the ecosystem service valuation worth highlighting, along with useful insights to direct future application of the approach used in this research, and future research on reactive nitrogen in the LYRB.

While the limitations in the land cover analysis have been discussed, the simplification process for land cover undoubtedly affected the accuracy of the valuation results. Ultimately the process of land cover simplification was driven by the availability of ecosystem service valuation studies at the simplified land cover precision, so errors resulting from the simplification of land cover can be attributed to a lack of either regional or land cover specific ecosystem service valuation studies. If the existing literature on ecosystem service valuation contained more detailed valuation studies on specific land covers, the accuracy of the benefit transfer would be greatly increased, as current limitations are due to valuation studies and not the precision of land cover data. If valuation literature was to increase in it's availability, perhaps this situation may change, but with the increasing availability of high-resolution satellite data, valuation data is likely to remain the limiting factor in this approach for some time to come.

Even with the ecosystem service valuation literature available, the definition of ecosystem services valued with clear documentation of specific ecosystem services included is lacking. When using primary or meta-analysis based ecosystem service valuation studies, if it is unclear the specific services valued, the results of a benefit transfer approach may inform a general understanding of ecosystem service values (as was the goal in this study) but the quality of valuation data will make it difficult for results to be seen as robust and policy relevant. To address these limitations, future primary valuation studies of ecosystem services need to be extremely clear as to the specific services valued.

Fortunately most of the studies used for the benefit transfer in this research did document the specific ecosystem services valued (Troy and Wilson 2006 did not document these for their CA and MA study sites), however this did not resolve all potential issues that may affect the accuracy of valuation. As shown in Section 4.2 Table 3, very few ecosystem services for a given land cover were valued by all of the six studies used as the basis for benefit transfer in this research. Forest, Grassland and Wetland land cover types were fairly well represented in each of the studies, whereas Pasture, Shrub/Scrub and Lakes/Rivers were consistently only valued by one to three of the studies. The emphasis on particular land cover types may be the result of locations where valuation studies were performed, but it may also indicate a lack of valuation literature on particular land cover types. This gap in the valuation literature may severely limit the benefit transfer approach for preliminary valuation of ecosystem services. Even more concerning for the LYRB study site is that some of the land cover types that are poorly represented (Shrub/Scrub, Pasture) are predominant land cover types in the LYRB and therefore any attempt to value ecosystem services in the LYRB must be focused on properly valuing the ecosystem services associated with these land cover types. Finally, a limitation of the studies used for benefit transfer is that as seen in Section 4.2 Table 3, not all land cover types provide all categories of ecosystem service. In some instances, this may be caused by the reality that certain land cover types simply do not generate all types of ecosystem service benefit, but the lack of valuation does not preclude the possibility that certain ecosystem services from various land cover types are actually being delivered, but because they have not been studied or published, they are not reflected in current literature. Clearly ecosystem service valuation is an evolving field, but it still important to highlight these limitations to address their relevance to this research and to drive improvements in future work.

While documenting specific services valued is a necessary first step to thorough ecosystem service valuation and may present limitations to accuracy, the current categorization of ecosystem services as shown in Section 4.2 Table 3, also may not provide a framework to accurately value ecosystem services and account for human capital investments in ecosystem service valuation. In the case of agricultural lands, only one of the six studies used provide a value for the provisioning service of food production, and very few studies attempt to value regulating services (gas, climate, soil, water) for this land cover type. Whether this a reflection on literature availability or on the meta-analysis performed is unclear, but the ecosystem service of food production on agricultural land requires human capital investment and it is unclear how this may or may not be accounted for. Additionally, regulating services such as carbon sequestration undoubtedly occur on agricultural lands, but this service is only valued in two studies (both conducted using the same analytical model). An additional component of the conceptual ecosystem service framework shown in Section

2.2 Figure 1, ecosystem dis-services, is not mentioned in any of the valuation studies used, and therefore the handling of this in ecosystem service valuations used in the benefit transfer is unclear. What these limitations highlight is that the methodology for defining and categorizing ecosystem services is not uniform, and therefore when attempting to use valuation data across multiple studies it is a challenge to define the specific services valued and how human capital investments and ecosystem dis-services may impact ecosystem services values. There is a clear need for a transferable methodology for ecosystem service valuation to improve the accuracy of benefit transfer research such as conducted in the LYRB, and to meaningfully incorporate an ecosystem service valuation approach into decision making processes.

Aside from the potential limitations in the definition of ecosystem services and the valuation literature used for the benefit transfer, it is also important to understand how the results and valuation statistics may or may not accurately express the ecosystem service values in the LYRB. Because generalization errors in point benefit transfer are nearly unavoidable even with a high level of correspondence between the study and policy site, benefit transfer for ecosystem service valuation seeks to minimize these errors as much as possible. Because the correspondence between any specific spatially explicit ecosystem service valuation study and the LYRB was low, the decision was made to combine values across multiple studies and accept the generalization errors that comes with this process rather than accept a known large generalization error created by transferring values from a distinctly different site to the LYRB. Because the goals of this research were to understand general values and particularly the relative values between ecosystem services from different land covers, rather than generating the most robust total estimate data, by combining values from multiple sites at multiple scales it is assumed that there are general relationships of relative value between ecosystem services at different land cover types that can be determined. Clearly, this approach does not define the exact value of a specific land cover in the LYRB, but it does illustrate likely relationships between values of land covers present, and their spatial distribution.

When using the values of multiple sites consideration was given to using average, median, minimum and maximum values to create high/low estimates. From examining the studies, it was apparent that certain land cover types were not well represented across all studies (shrub/scrub) or that estimates for a given land cover demonstrated a large range between high and low values. Because of this wide range, minimum, maximum and average values appeared to be significantly influenced by the extreme high and low estimates and therefore represented values from one particular study disproportionately. By using median values, these effects were minimized, as well as providing a more conservative estimate of value. This decision may have affected the total ecosystem service value estimates, but it impacted the general relative values of ecosystem services by land cover type to a much lesser extent.

Undoubtedly there may be concerns raised about the accuracy of the total annual ecosystem service values generated, and while some of these concerns may have already been addressed in this section, the interpretation of the total ecosystem service values is an important component of this research. The range for annual low and high median estimates ($554,320,594 - $3,996,211,899) is large, and correctly indicates a large degree of uncertainty. However, this uncertainty is expected when considering the importance of ecosystem health and productivity and characteristics of study site population and land use. The land cover approach to ecosystem service valuation does not include information regarding the actual condition of ecosystems, and therefore there may exist wide differences in ecological productivity between lands classified as a given land cover type. Additionally, a deciduous forest may provide a very different set and scale of benefits from an evergreen forest, yet these were valued the same in meta-analysis or benefit transfer. Because of this it is unavoidable for there to be a range of potential values, and narrowing this range or identifying a single specific value would require a more in depth understanding of ecological health and productivity across a study site. Additionally, ecosystem services are defined by the benefits received by humans, and therefore without accounting for site specific population characteristics, a wide range of ecosystem service values is likely. An extremely healthy and productive ecosystem far removed from human populations will likely have less ecosystem service value than a less productive ecosystem in closer proximity to human populations that use the ecosystem for recreation, water treatment or materials. Because the benefit transfer approach does not account for this level of regional specificity, a wide range of estimated values is to be expected.

So while there are many limitations to the benefit transfer results, and a wide range of uncertainty due to the lack of regionally- specific data on ecosystem health and human land uses, there are still valuable conclusions to this valuation methodology as applied to the LYRB. Clearly, the results of the preliminary valuation are not robust enough to use in economic decision-making or in comparative analysis of land uses and economic development in the LYRB, but the results do provide useful estimations of relative values and begin to generally illustrate potential differences in magnitude of ecosystem services and their distribution. While deriving ecosystem service values from multiple sites generated generalizations errors, it also increased the validity of underlying general relationships between the values of ecosystem services delivered by different land cover types. The impacts of human populations and ecosystem productivity are critical to precise ecosystem service values, but without the ability to include these components with the multiple goals of this research, a general estimation of ecosystem service benefit distribution and magnitude was all that was desired. In a similar manner to the assumption that underlying value relationships exist to validate meta-analysis valuation techniques, in this research it is assumed that there exists an underlying general relationship between ecosystem service values for different land cover types. By incorporating values from multiple sites, this underlying relationship may be more adequately reflected than by selecting values from a single study site for point benefit transfer.

A component of the ecosystem service approach, which is surely a point for valuable discussion, is whether attributing dollar amounts to non-market ecosystem goods/services is a valid approach to inform land use decision-making. Market goods and services are already being associated with a given economic measure of value, so are irrelevant to this discussion, but non-market goods it may be argued, are not accurately represented in economic figures. Clearly, a definitive answer to this argument is beyond the ability of this research, but from the results of this research we can see that even a very generalized preliminary ecosystem service valuation can assist in locating and identifying areas likely to provide important services. If the dollar figures are removed and value is assessed in a rescaled relative value index this same objective is achieved. So while assigning specific dollar amounts may imply a basis for comparing ecosystem goods/services to other goods/services, understanding the context of the dollar value estimates for ecosystem services in this study is critical to understanding if such a comparison can be justified. In the case of this research, the dollar values are not likely to provide a valid basis for comparing ecosystem services to other goods/services, but that does not mean the distribution and relative value results are not useful for understanding ecosystem services in the LYRB. While dollar figures do present an opportunity for comparative or cost-benefit analysis, it is not necessarily appropriate to do so, and understanding these potential limitations may reduce concerns over the validity of using economic measures for ecosystem service valuation.

In addition to highlighting the limitations and justifications for economic methods used in the benefit transfer, it is critical to highlight results of this component of the research and how they can inform decision-making in the LYRB. The valuation data clearly supports the importance of wetland ecosystems in the LYRB, and indicates that future research on understanding ecosystem services in the LYRB should likely focus on a more complete valuation of these ecosystems. Because of the large area in the LYRB, shrub/scrub ecosystems play an enormous role in regional ecology and because this land cover type is not well represented in the existing ecosystem service literature, this is another component in need of future research highlighted by this study. In terms of land use planning, changes in wetland or forest areas should likely consider the potential for benefit losses/gains through ecosystem services. Agricultural ecosystems, while represented as between 6.3-6.4% of total annual ecosystem service value, have numerous difficulties in terms of interpreting these values because of the presence of significant human capital investments in these ecosystems. Until a more thorough valuation methodology that specifically addresses this reality is incorporated into ecosystem service valuation, it is difficult to draw any strong conclusions for decision-making.

Determining the distribution and preliminary values of ecosystem services in the LYRB was a clear goal of this research, but in addition to this it was hoped that this research would serve to inform how a preliminary ecosystem service valuation approach could be completed with limited resources and what the expected outcomes of this approach would yield. This research demonstrates that a preliminary valuation can be completed using a benefit transfer and land cover approach, but that the valuation will likely only be general enough to yield information on the relative values and distribution of ecosystem services in a study area, rather than generate robust economic metrics. If this approach was applied to another study site, there is the potential for increased reliability of valuation data if a study site with a high level of correspondence to the policy site is located, but unfortunately this was not the case for the LYRB. Additionally, due to the relatively small volumes of literature conducting spatially explicit ecosystem service valuation, the chances of finding sites with high correspondence is low. Because of this, research using this method of preliminary valuation should focus on realistic goals of defining general ecosystem service value distribution and increased understanding of relative values within a study site.

A significant value of this research approach is also to direct primary ecosystem service valuation (using stated or revealed preference methods) towards specific regions or land cover types which may be of the most concern due to highest ecosystem service values, human land use or both. In the case of the LYRB, the preliminary valuation would direct future work towards wetlands, and shrub/scrub as mentioned earlier, however the exact areas of focus in future applications of this approach would be dependent upon study site characteristics. In terms of scale, this approach could be applied to regions both larger and smaller than the LYRB, however data quality and resolution may become more significant issues as study site scale changes. A final critical point is that while the preliminary valuation was done without direction from the actual study site population, using only satellite data and academic literature, any future ecosystem service valuation work in the LYRB should receive input and data from the study site population and should be largely directed by the local population. Because ecosystem services values are inherently built upon human values, a solid understanding of population values and characteristics will be imperative to create policy and regionally appropriate studies of ecosystem service values.

Discussion Section 5.3: Nitrogen Loading Estimates

Before analyzing the ecosystem service valuation results with the estimated landscape nitrogen loading data, it is necessary to examine the nitrogen loading data results for a complete perspective on the subsequent analysis. One of the most important aspects of the nitrogen loading data is that it is estimated, modeled data, and is based upon assumptions and simplifications that may contain differences between actual nitrogen loading conditions. Understanding the assumptions inherent in these data provides a more complete context for interpreting research results and an opportunity for understanding how the modeled data may be improved to more accurately represent actual physical nitrogen loading conditions.

Beginning with the CAFO loading layer, the primary assumption driving loading estimates from these sources is that animal wastes generated will be applied at greater rates closer to these facilities, and will only be applied on agricultural land covers. While this assumption is based on management practices used by these operations, because detailed waste management plans are regarded as trade secrets, assumptions of waste management are all that is currently available to scientists modeling the potential environmental impacts of these facilities. To address concerns over potential errors in modeled loading data, increased access to CAFO waste management practices would be an extremely beneficial step to improve confidence in these nitrogen load sources. As another major potential source of nitrogen loading, the FARM loading layer may contain even greater assumptions than data generated for CAFO sources. The FARM layer attempts to represent nitrogen loads from fertilizer inputs, and simply used 2002 data on total county fertilizer use, and distributed this fertilizer equally to all cultivated crops and pasture land. This is an obvious over-simplification of the actual nitrogen loading conditions, and is likely to underestimate particular areas of intense nitrogen loading due to different nitrogen requirements and biological nitrogen fixation rates, and overestimate in other areas with less intensive fertilizer use. The CROP layer attempts to represent the same fertilizer use information in the FARM layer by estimating fertilizer use based on crop types grown according to the USDA Crop Data Layer, however because of the limited extent of this layer, it was not used in this analysis. Examination of the FARM loading data appears that it estimates nitrogen loading conservatively, but given the methods of equal application across the entire study area this to be expected.

Assumptions in the NONFARM, PASTURE and RANGE layers also took total regional estimates and applied them equally to a relevant land cover type and topographical area, and these also likely created areas of estimated loading error. The NONSEWER layer assumed even population distribution for septic sources on developed lands, which likely missed areas of concentrated populations and increased nonsewer loading. Such hotspots of nitrogen loading from septic systems have been found in the LYRB (Parker, WA) and are not reflected in the nitrogen loading data. Finally, the ATMOSPHERIC layer was based completely upon the CMAQ model, therefore uncertainties and errors in the construction and output within this model must be consulted to appropriately understanding limitations in this data layer.

Even with the limitations and assumptions of the nitrogen loading data, there are still significant results that appear dominant enough to remain true even under additional refinements to models and added management information. Specifically, the strength of CAFO and FARM layer nitrogen inputs was so great that even with improvements to model estimates, these sources are still very likely to be the most significant in the LYRB. Also, due to the location of these sources, this analysis shows with a level of confidence that the most intense nitrogen loading in the LYRB occurs in the basin floor, and specifically in the Lower Yakima Valley, roughly approximate to the GWMA boundaries. While the exact loading rates may contain errors based on model assumptions, the locations of these sources and their likely role as major sources of nitrogen are still quite clear in the estimated loading data.

In order to generate more accurate nitrogen loading data there are several key components of the nitrogen loading layers that should be addressed. First, gaining access to CAFO and dairy operator's waste management plans would dramatically improve the estimates of nitrogen loading from these sources. Detailed application rates and locations of wastes would allow a much better estimation of the contribution of these sources to total nitrogen loading in the LYRB, and would assist in estimating fate and transport processes on reactive nitrogen in the study site. Second, connecting specific crop types with nitrogen loading data would improve the accuracy of the FARM estimates by connecting actual land use activities with estimated nitrogen loads. Gaining access to fertilizer management plan by local farmers would also be a valuable method for gaining higher quality nitrogen loading estimates. Third, obtaining data on the LAND applied nitrogen layer from wastewater treatment facilities would allow for the mapping of these sources as polygon rather than point features, which would allow these sources to be included in the analysis of this research as well as future nitrogen research in the LYRB. Finally, mapping actual septic units in the LYRB and connecting information regarding the size, age, condition of these systems would increase the validity of nitrogen loading estimates for the NONSEWER layer.

Discussion Section 5.4: Nitrogen Loading and Ecosystem Services

Spatially explicit ecosystem service using benefit transfer and techniques for estimating landscape nitrogen loading are well represented in existing literature, but connecting these two components was one of the most unique contributions of this research. While much of the statistical and spatial analysis results are unique to the LYRB study site, the lessons learned from this research approach are valuable for identifying suitable applications of this methodology and understanding how results may be used to inform management and regulations to protect human and environmental health.

An initial goal in connecting ecosystem service value estimates with nitrogen loading data was to gain a better understanding of the potential relationships and opportunities for metrics for use to maximize benefits from ecosystem services and reactive nitrogen while minimizing any benefit loss due to excess nitrogen inputs. The first step in this was summarizing estimated ecosystem services by land cover and the nitrogen loads experienced (see Figures 17 & 18). What we can see from estimated nitrogen loads and ecosystem service values estimates by land cover is that land covers in the LYRB demonstrate a variety of relationships, and cannot be assumed to function in the same relationship across different land cover types. While in some instances high nitrogen loading may result in decreased ecosystem service benefits, this cannot be assumed due to the ability of ecosystems such as wetlands to serve as nitrogen sinks (Jordan et al. 2010) or areas of denitrification, which would actually increase the ecosystem service benefits. Defining this relationship therefore requires a site-specific inventory of ecosystem productivity and human use. What may be helpful in furthering the understanding of these relationships would be the identifying of estimates of potential maximum nitrogen loads before negatively impacting ecosystem productivity that could be applied to LYRB estimates to understand the potential for increasing ecosystem services through reactive nitrogen management. However, even if nitrogen thresholds for ecosystem productivity functions were identified, the issue of defining ecosystem services as addressed in the conceptual model for an ecosystem service framework (see Figure 1) would still need to be addressed to appropriately account for ecosystem services and dis-services.

Moving on from the relationships of magnitude between ecosystem service values and landscape nitrogen loading, it was hoped that this research would be able to identify trends in spatial distribution of these properties in the LYRB. The ability to identify areas of concern due to especially high nitrogen loading, high ecosystem service values or both, would clearly be beneficial to focusing regulatory efforts and management decision-making and was an important goal of this research. The findings that there was not a strong correlation between nitrogen loading and ecosystem service values at the individual polygon scale, was to be expected, given the fact that CAFOs and areas of cultivated crops are rarely located in the exact same location as land cover with high ecosystem service values such as forest or wetland. However, moving to a more generalized scale for HUC 12 and HUC 10, it was anticipated that correlation would become stronger and significant in the LYRB. Because agricultural activity and CAFOs are predominantly located along the valley floor, it was expected that the areas with high ecosystem service values along the Yakima River would be sufficient to show a statistically significant correlation. It was hypothesized that while nitrogen loading may not occur in the *same* location as high ecosystem service values are generated, nitrogen loading does tend to occur *near* areas of high ecosystem service. However, at the HUC 12 and HUC 10 scale, this correlation was not strong enough to adequately confirm this hypothesis. It is important to note that this result was a feature of land cover and land use in the LYRB and the same approach at a different study site, or even within a smaller area of the LYRB, may in fact confirm this hypothesis at a different scale. The benefit of demonstrating this correlation would be the ability to demonstrate statistically to local residents that human activities have real potential to reduce ecosystem service benefits and therefore should be an important issue and consideration in the regulatory and management discussions.

Despite the fact that strength of correlation between ecosystem service values and nitrogen loading could not be demonstrated in the LYRB, there were still significant results from the analysis that are important to highlight. Plotting ecosystem service values and nitrogen loading by HUC 12 and HUC 10 areas still was able to successfully demonstrate specific areas in the study site where high ecosystem service values and high nitrogen loading both exist. What this demonstrates given the quality of data in this study, is that these areas may be of particular concern for reductions in ecosystem service benefits. Because the valuation data assumed a general level of ecosystem health and productivity, high nitrogen loads may actually be significantly reducing the ecosystem service benefits in these areas. On the other hand, these areas could also be significantly undervalued if ecosystems in these regions are performing critical biogeochemical functions that actually minimize the impacts of nitrogen loading, especially in terms of aquatic ecosystem services. In terms of the LYRB, this level of analysis specifically highlighted the Lower Yakima Valley, roughly represented by the GWMA boundaries, as the area of most concern. While this is not new information to land managers, residents and regulators in the LYRB, confirmation of the importance of this area using an ecosystem service approach serves to strengthen existing efforts to address concerns over reactive nitrogen in the LYRB. In areas without the history of nitrogen awareness in the LYRB, this research approach could serve a primary role in identifying areas of concern, rather than simply a confirmation of existing work.

It is important to include in the discussion of this research how an ecosystem service approach and the methods used in this research may be applied to the issue of reactive nitrogen management and what anticipated uses the results of this approach may yield in terms of regulatory and land use management decision-making. In analyzing the estimated ecosystem service values and nitrogen loading data, information on the spatial correlation between these characteristics can be important to serve as justification for considering ecosystem services in the land management process. As demonstrated in this research, the strength of this correlation is not imperative to useful results, but demonstrating this relationship can be beneficial to communication of the importance of addressing nitrogen in a given area. In addition to tests for correlation, this approach will identify areas with high ecosystem service values and high nitrogen loads, which will assist in directing stakeholders to areas of highest concern. In this research, these findings supported existing management priority areas in the Lower Yakima Valley, but if applied to a different study site or at a different scale, the approach may be of increased utility to stakeholders. Other potentially valuable outputs from the ecosystem service approach to nitrogen management taken in this research could be developed in conjunction with further research identifying ecological productivity functions and nitrogen thresholds for different land cover types. Such thresholds would assist in estimating the affects of current nitrogen loading and maximizing ecosystem services by increasing under utilized biogeochemical processes or decreasing negative affects of excess nitrogen upon ecosystems unable to maintain optimal levels of productivity under current loading conditions.

The approach taken in this research can best be seen as a tool for preliminary investigations that will produce general conclusions to help focus future research efforts towards specific ecosystems and nitrogen loading sources. In the LYRB, the specific conclusions in this regard indicate the importance of wetland areas due to their proximity to areas of high nitrogen loading, particularly in the Lower Yakima Valley and their high estimated ecosystem service values. Future research on ecosystem services in the LYRB would be well advised to focused on these particular ecosystems and refining measures of ecological productivity, ecosystem service benefit estimates and impacts of nitrogen loads received. This research also supports a focus on farm and CAFO nitrogen sources due to the magnitude and location of loads generated from these sources. While other sources of nitrogen clearly exist and may be of increased significance in particular areas in the LYRB, the areas of most intense nitrogen loading are clearly driven by farm and CAFO sources, justifying a focus on these sources for land managers and regulatory authorities. Ecosystem service value estimates for particular land cover types is also an area in need of potential refinements due to the lack of valuation literature (shrub/scrub) or high contribution to total ecosystem service value estimates in the LYRB (forest), but given that these land covers are removed from the areas of highest nitrogen loads in the LYRB, and they should be of lesser priority in order to address the most immediate potential impacts of nitrogen loading.

Discussion Section 5.5: Groundwater Ecosystem Services

A major finding of this research was the lack of information regarding groundwater ecosystem services in existing ecosystem service valuation literature. While a spatially explicit ecosystem service valuation approach may provide useful information regarding reactive nitrogen management, the relationship to groundwater ecosystem services is not one of the strengths of this approach. In the LYRB this is a very important consideration when attempting to apply an ecosystem service approach to this issue due to the prevalence of groundwater users and concerns over nitrate levels. In any study area in which this research approach is applied, it is clear that additional econometric methods or human health metrics must be employed in order to capture any of the potential impacts of nitrogen loading to subsurface ecosystem services. It is recognized that due to the importance of groundwater in the study area, the valuation of groundwater quality, specifically for nitrate levels will be a major point of interpretation and discussion. This section will provide a basis for interpreting the results of the benefit function transfer performed, and understanding how these results may be contextualized within an ecosystem service framework and with regard to LYRB population characteristics.

Because ecosystem services provided by groundwater resources and subsurface processes are not explicitly incorporated into ecosystem service valuation literature, it is necessary to define these services within the ecosystem services framework. Using the conceptual model for ecosystem services shown in Figure 1 (Section 2.2) groundwater can be seen to provide direct ecosystem services from primary ecosystems through regulation of water quality by subsurface processes, as well as direct ecosystem services through the provisioning of water supplies. There are also critical indirect services provided by groundwater in terms of relationships between ground and surface water that may assist in primary ecosystem services such as habitat, provided by surface waters. It is important to note that while it does typically require capital investments to access groundwater resources, these investments are not required in the actual generation of the ecosystem benefit, only in the subsequent human use of this benefit, so groundwater resources should not be seen as an engineered ecosystem benefit unless managed aquifer recharge or similar methods of engineering are employed to cultivate benefits from groundwater resources. Despite this, it is clear that groundwater resources may generate significant ecosystem services, and particularly in arid environments with a high reliance on groundwater resources such as the LYRB, it is absolutely imperative that these services are included in any examination of regional ecosystem services.

One of the most critical components to interpreting the results of the groundwater valuation in this research, is the recognition that the regression model used does not estimate total value for groundwater resources, but only estimates household willingness to pay for water within the maximum contaminant load (MCL) for nitrate-N defined by the US EPA as 10 mg/L (EPA 2012). While drinking water safety is likely a significant component to total groundwater value, it is not total value and therefore the estimates provided by the regression model used represent conservative estimates compared to total value. But because this research is focused on reactive nitrogen, understanding the relationship between nitrate level and the value of groundwater is primary a goal, and the regression model used was chosen specifically because it estimates this component of groundwater quality. In terms of ecosystem services, the regression model only estimates ecosystem services related to nitrate level, not total ecosystem services. Specifically, these value estimates are focused on the direct ecosystem services of nitrogen regulating subsurface processes, and the provisioning of drinking water from groundwater resources within the designated MCL. Indirect services such as nitrate level of groundwater used in irrigation or to support other engineered ecosystem benefits is not included in this valuation. These indirect services may be significant, but were not estimated in this research.

The updated WTP estimates from the Crutchfield et al. 1997 regression model indicate that total estimated WTP for all the census tracts included is \$57,526,003 annually (\$4,793,834 monthly) with household annual WTP varying between \$613.12-\$910.72 (\$754 average) depending upon the specific census tract. While these numbers may be clearly understood to those well versed in economics, to regulatory agencies and other stakeholders, the interpretation of these figures may require further explanation. The WTP estimates the value of groundwater to remain within the MCL for nitrate, and therefore serves to inform the benefit loss generated by water above the nitrate MCL on a household level. Because even households that do not directly use groundwater from private wells as their primary drinking supply may still value groundwater quality in the study area, this population is also included in value estimates. The regression model estimates that if all groundwater in the census tracts included was above the nitrate MCL, the average benefit loss would be \$754 annually to all households in the area. Using current estimates of wells with groundwater above the MCL

(PGG 2011), the current annual benefit loss in the LYRB is estimated to be $720,161. Given that the US average water bill is estimated at $300 per household annually (EPA 2004), the benefit loss due of $754 per household annually due to elevated nitrate levels in the LYRB can be seen as approximately 2.5 times the average annual cost across the country. While well users investments may be very different than those experienced by users of municipal water services, for certain populations such as renters, a comparison of monthly costs is relevant. The WTP estimates should not serve to inform regulatory or management agencies directly as to what residents may be willing to spend to see improvements to groundwater quality, but it does provide estimates of benefit loss to local residents from elevated nitrogen levels which should inform comparative analysis and management decision-making.

With the WTP estimates, it is also useful to consider these values within the concept of ecosystem service values from the preliminary valuation and in the context of population demographics in the study area. While the ecosystem service value estimates from this study are intended as general preliminary estimates, and issues using these for comparative analysis have been discussed in Section 5.2, if we compare the total benefit loss potential due to nitrate levels in groundwater exceeding the MCL, we find that using low median estimates for ecosystem service value this benefit loss is lower than ecosystem services delivered by wetlands, forest and shrub/scrub lands, but greater than any other land cover type in the LYRB. In terms of total ecosystem service value, ecosystem services relating to nitrate-N levels in groundwater represent 9.4% of total ESV (low median). Clearly, the accuracy of these numbers for comparative analysis is a concern, but the relatively high value of maintaining safe nitrate levels in groundwater for the LYRB does provide another indication of the importance of groundwater services and nitrate levels in the LYRB.

To relate the estimates of groundwater value and nitrate levels to other components of this research, it is helpful to understand population distributions in relation to nitrogen loading and generalized groundwater recharge and flow estimates. Because areas of higher groundwater recharge may indicate the potential for increased movement of reactive nitrogen into the subsurface, it is useful to define the estimated groundwater recharge conditions in the LYRB as shown in Figure 37 (Ely et al. 2011). As shown in this figure, while the Lower Yakima Valley is estimated to have the highest recharge rates due to both hydrogeology and land use factors such as irrigation, the elevated recharge rate also indicates a greater potential for nitrogen movement into groundwater in this region. The magnitude of nitrogen movement in this area would need to be confirmed through additional analysis, but these recharge estimates support at the very least a concern for elevated nitrate levels, and increased benefit loss in this area. Connecting recharge in this region with nitrogen loading estimates from earlier components of this research, and estimates of generalized groundwater flow, we get a more complete perspective of nitrogen, groundwater and population centers as shown in Figure 38 (Generalized groundwater flow data: PGG 2011). Considering the information from both Figures 37 and 38, it is apparent that the Lower Yakima Valley area contains the highest groundwater recharge in the LYRB along with the areas of highest landscape nitrogen loading. The generalized groundwater flow patterns in this area also indicate that the population centers of Granger and Sunnyside may also be most at risk for experiencing elevated nitrate levels in groundwater due to the estimated hydrologic flow paths that connect these populations to areas of high nitrogen loading upgradient.

Connecting the likelihood for elevated nitrogen movement into groundwater in the Lower Yakima Valley and the potential for this nitrogen to flow towards population centers, it is clear that residents in this area may experience higher benefit loss in terms of groundwater ecosystem services than in other parts of the LYRB. While the potential for increased exposure to health risks from elevated nitrate levels or increased potential for benefit loss may be a reason for concern from regulatory agencies and land managers, it requires further research to identify if environmental justice considerations are warranted. In order to demonstrate the existence of environmental justice issues a detailed analysis of regional populations and land uses would need to be performed, which is beyond the scope of this research (for a review of this type of analysis see Bowen 2002). However, as seen in Figure 39, the percentage of the population living below the poverty line in the LYRB can provide greater context for understanding the potential for environmental justice concerns. Given the national average of percentage below the poverty line of 15.9% (Census 2010), income estimates in the LYRB demonstrate above average poverty rates, particularly in urban centers and along the basin floor. The areas of Sunnyside and Granger, which have been highlighted for their risk for increased nitrogen loading to groundwater, demonstrate poverty rates 9.2-19.1% above the national average and therefore indicate the potential for populations facing economic insecurity to be exposed to the greatest benefit losses from nitrogen loading, and potentially increased health risks. Definitive analysis of environmental justice issues is beyond this research, but clearly the nitrogen loading estimates and groundwater benefit loss potential from nitrate levels as demonstrated in this research supports the inclusion of environmental justice issues in the discussion of reactive nitrogen in the LYRB.

Finally, it is also necessary to address the potential limitations of the application of the Crutchfield model for estimating groundwater ecosystem services. One potential area of concern may be correspondence between the study area in which the original valuation was performed and upon which the regression model is based and the LYRB. Because the original study area of the Mid-Columbia Basin actually includes the LYRB, the level of correspondence is high, and it is not likely a source of significant error. Additionally, while there is uncertainty embedded in Census measurements of population characteristics, this represents the best possible data available (short of primary data collection) and was not able to be addressed within this research. The most likely sources of difference between the original econometric model and its application to the LYRB are assumptions for model variables for which updated information was not available. Specifically the variables for rural living, connection to municipal water system, awareness of nitrate issues, use of treatment system, use of bottled water and years at zip code used either mean 1997 values provided by Crutchfield et al. 1997, or were altered at the discretion of the author. Rural living, connection to municipal system and awareness of nitrate issues were all updated based upon spatial characteristics of census tracts and current conditions in the LYRB. While it may be argued the values selected for these variables are without adequate justification, these values were consistently chosen to be conservative estimates, and therefore likely undervalue WTP estimates. Future primary research on population characteristics would result in a more accurate application of the Crutchfield regression model, and would be an area for consideration if higher precision WTP estimates are desired at either a regional or census tract scale.

Figure 37: Mean Annual Recharge (2011 Conditions)

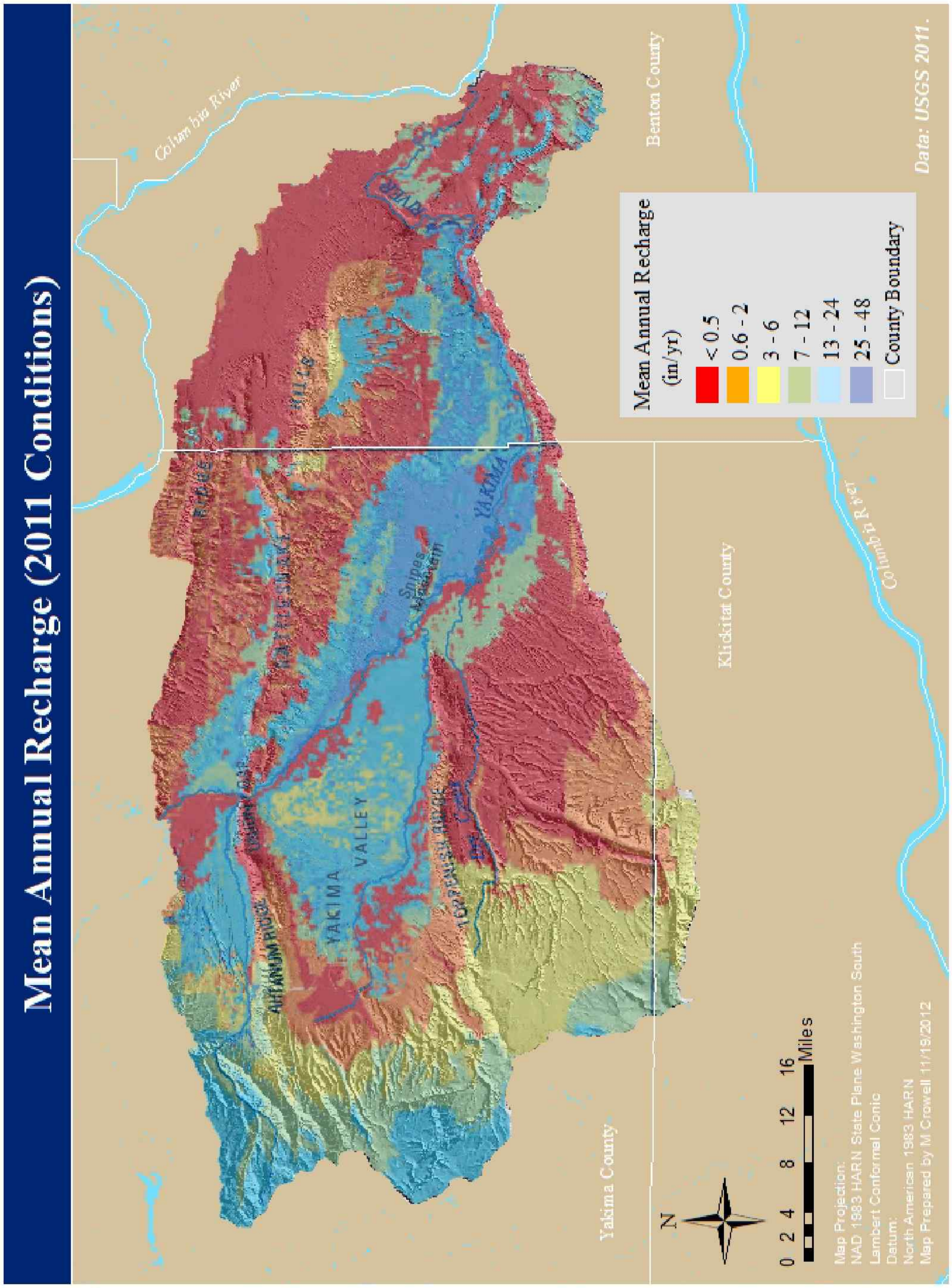


Figure 38: Estimated N Loading and Generalized GW Flow

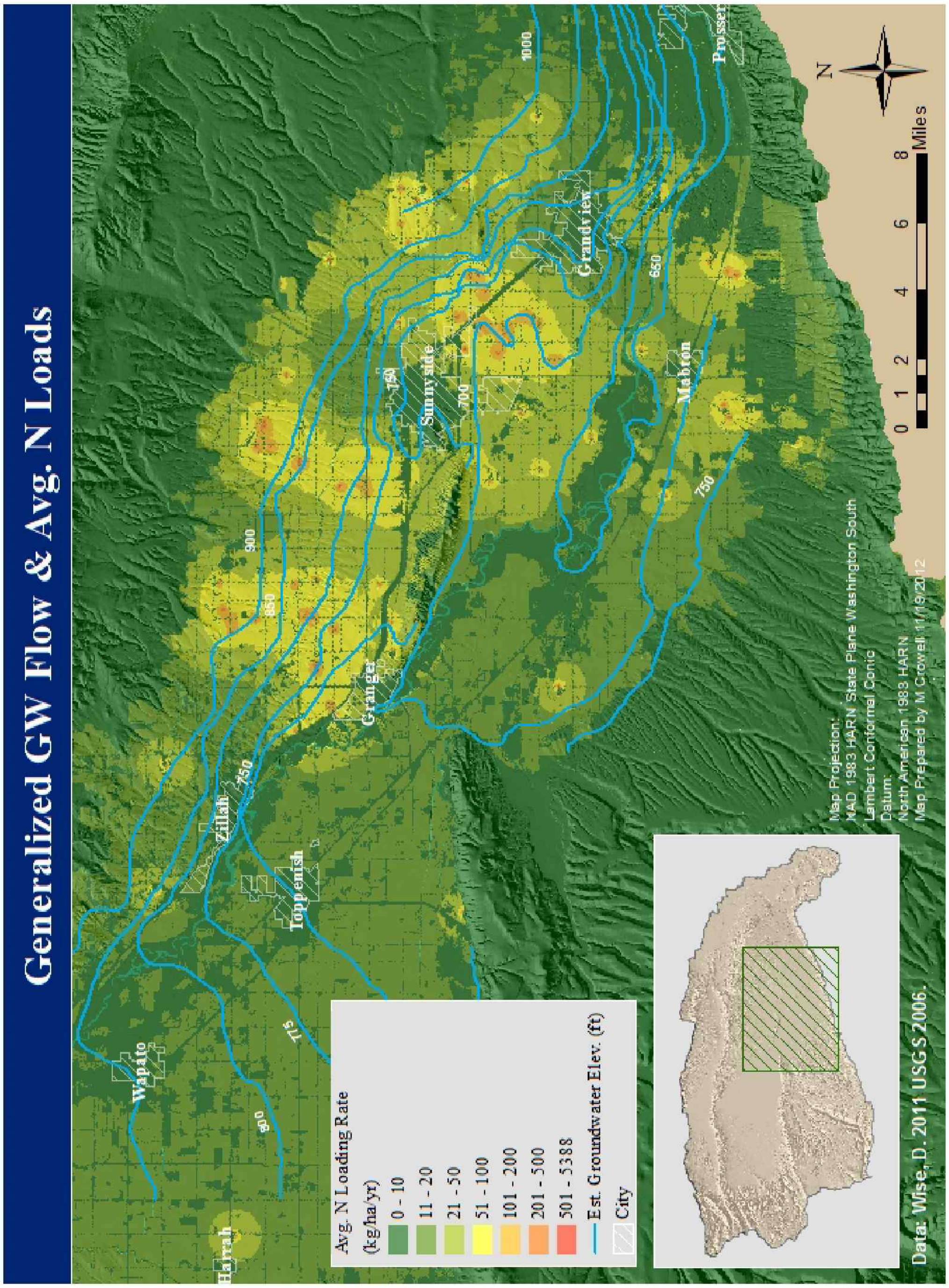
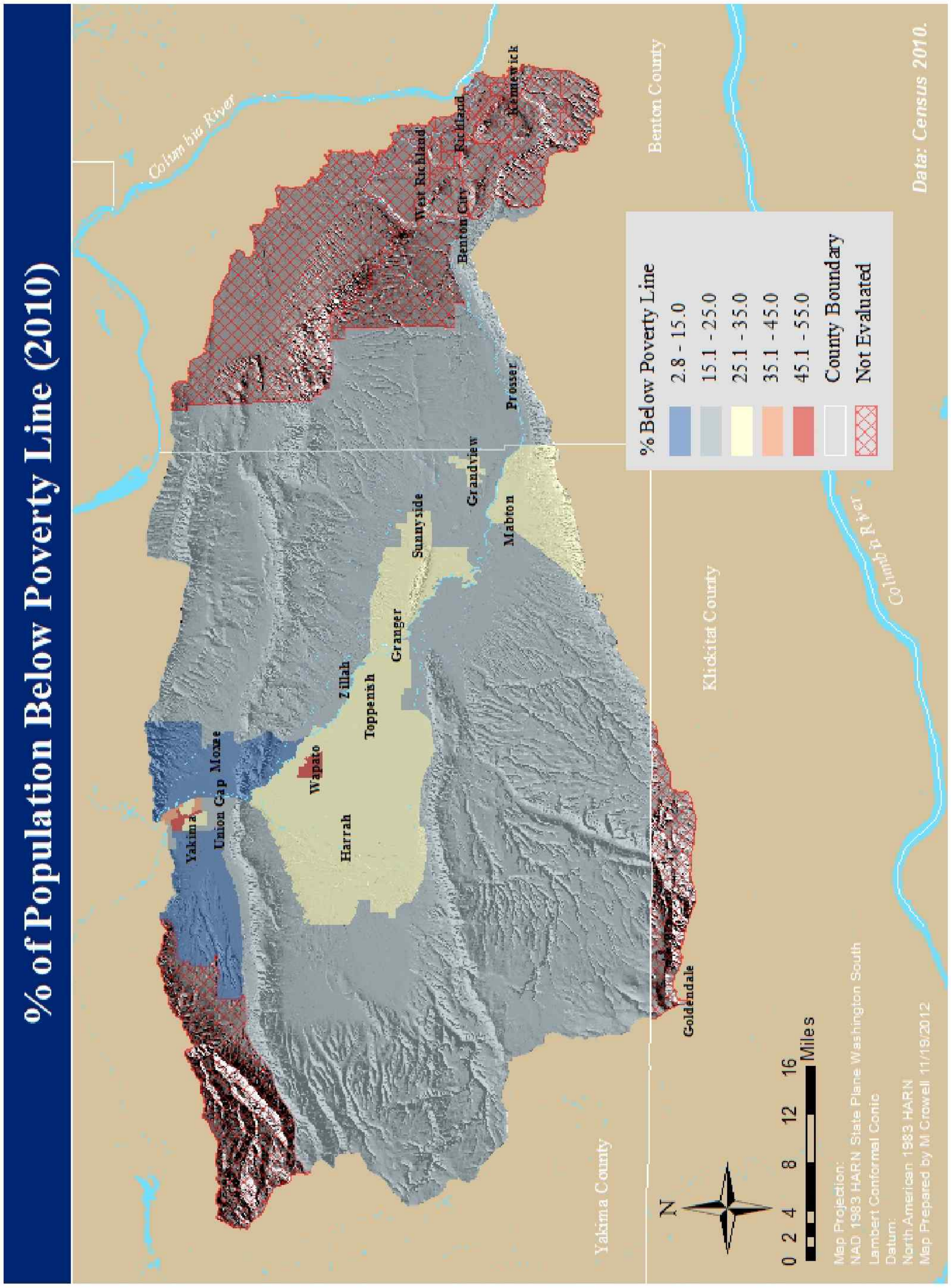


Figure 39: % Below Poverty Line



Section 6: Conclusions

The application of an ecosystem service framework to the issue of reactive nitrogen management provided a valuable opportunity for detailed analysis of the ability of this framework to address specific environmental issues. While some of the conclusions of this research are specific to the LYRB study site, there are a variety of findings that will assist in future development and improvement of the ecosystem services approach and its application to environmental issues such as nutrient management.

In terms of contributions to the development of ecosystem services as a tool for application, this research clearly highlighted the general level of conclusions which should be anticipated using the ecosystem service approach. A finding of this research is that an ecosystem service is approach is currently limited due to differences in ecosystem service typologies and classifications. Without standards for defining ecosystem services and benefits it is not possible to ensure that human capital investments in generating ecosystem benefits have been properly incorporated, and that direct and indirect services are not subject to being neglected or experiencing counting errors in valuation. The understanding of exactly what ecosystem services are valued is a first step in translating existing literature for use in a benefit transfer approach, yet these standards are currently lacking. Because of the lack of standards across studies, at this time a benefit transfer ecosystem service valuation approach can only be expected to generate valuation data that is extremely general and with a wide range of values. Even with standardized typologies for accurately valuing ecosystem services a wide range of values is still unavoidable because of the assumptions of human use and ecosystem productivity that made when using a land cover approach that does not specifically inventory regional activities and ecosystem conditions.

Accounting for these limitations, the ecosystem service approach taken in this research does demonstrate that even without the highest level of precision in total value estimates, there can still be useful conclusions drawn from the relative values of ecosystems in a study area that can inform future research and regulatory focus. Specifically, estimating the distribution of ecosystem service values can inform management priorities even without precise economic data. As done in this study, patterns of ecosystem service values can be compared to environmental concerns such as nutrient loads, contaminant pathways, etc. which allows for the assessment of potential benefit loss from reduced ecosystem services. This application of an ecosystem service approach indicates that this may be a useful tool for incorporating ecosystem services into spatial analysis and it is envisioned that future work will continue to build upon this methodology. It is extremely important to note that the ecosystem service framework does not successfully integrate subsurface ecosystem services and requires additional analysis to do so. This should be seen as a major shortcoming of this approach in areas where these resources have considerable utility to local ecosystems and generate human benefits.

Another critical benefit highlighted in this research is that the integration of a preliminary valuation of ecosystem services with environmental concerns such as nitrogen management can be done within a relatively small budget and time frame using available spatial data and generate useful results. Understanding the potential limitations of this approach is critical, but with these in mind, the approach taken in this research could easily be replicated in other study sites at a variety of scales for any number of environmental issues. The transferability and relatively minimal investments to conduct this type of assessment allow this approach to potentially be used by a diverse variety of stakeholders for a wide range of issues. Lastly, a general conclusion of the ecosystem service approach used in this research is that the ecosystem service framework does allow for the incorporation of multiple benefits for comparative analysis. While there is still uncertainty in the valuation literature on defining and categorizing these benefits, it is a clear asset of this approach that multiple benefits from individual ecosystems are incorporated into measures of value.

In terms of the specific application conclusions for the LYRB study site, this research clearly indicated that there is increased potential for reductions in ecosystem services due to nitrogen loading specifically in the Lower Yakima Valley. The occurrence of both high ecosystem service values and high nitrogen loading rates demonstrates the potential for benefit losses due to excess nitrogen. Quantifying the actual and potential damages from nitrogen loading in this area will require physical monitoring of this area, but this research should inform future efforts and support existing work in this region. Additionally the potential importance of wetland ecosystems is clearly shown, and these areas should be prioritized for future study or environmental protection efforts.

The sources of nitrogen loading estimated in the available data also indicate with considerable strength the importance of CAFO and farm source in nitrogen loading in the LYRB. While there are an abundance of nitrogen sources in the LYRB, the disproportionate loading from CAFO and farm sources presents a strong indication that these should be a primary focus in addressing concerns over reactive nitrogen management. These sources are even more dominant in the Lower Yakima Valley, which in addition to being highlighted for high nitrogen loading and high ecosystem service values, may also be the region with the greatest public health risk. Specifically considering the hydrologic factors relating to groundwater recharge and flow paths, protecting groundwater quality in this area should be a priority the land use managers and regulatory agencies. Failure to protect groundwater quality in this area puts significant ecosystem services provided by groundwater resources as risk of considerable benefit loss that will be experienced by all residents in the region.

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Appendices

Appendix 1 – All Land Cover Values from Valuation Studies (2012 USD)

	Troy and Wilson 2006					Liu et al. 2010	Costanza et al. 1997	Qenani-Petrela et al. 2007	Batker et al. 2011		Schmidt and Batker 2012	
	Massachusetts		*Maury Island, WA*		*CA Counties*							
	low	*high*	*low*	*high*					*low*	*high*	*low*	*high*
agriculture					$2,674.24			$2,650.87	$232.17	$1,699.88	$204.94	$1,589.06
beach			$93,959.52	$121,257.02		$127,059.96						
beach near dwelling			$114,684.88	$171,416.10								
coastal							$6,280.60					
coastal riparian			$6,761.24	$16,162.56								
coastal shelf						$1,869.11	$2,495.50					
conifer forest					$1,001.62			$993.19				
coral reefs							$9,416.25					
cropland	$4,180.94	$4,180.94				$69.34	$142.60					
disturbed and urban	$0.00	$0.00										
estuary					$7,195.56	$2,155.50	$35,389.60					
forest	$1,348.10	$2,963.38	$623.42	$3,833.24		$3,867.84	$1,501.95		$1,049.09	$9,632.02	$973.10	$14,544.21
fresh water bodies (lakes/rivers)	$193.98	$46,583.26				$2,306.23	$13,171.90	$8,752.17	$90.24	$60,352.52	$90.24	$9,489.39
freshwater stream			$1,145.58	$1,945.90								
freshwater wetland	$10,338.28	$46,563.74	$40,195.34	$117,235.90	$13,387.06	$26,212.69	$22,916.75	$13,270.64	$1,783.42	$91,381.69	$28,694.77	$154,024.06
grassland/herbaceous			$143.96	$143.96		$36.18	$359.60		$362.28	$1,820.26	$362.01	$1,820.26
hardwood oak woodland					$535.58			$531.40				
marine							$894.35					
mixed forest					$1,007.72			$998.68				
nearshore aquatic habitat			$1,041.88	$2,405.84								
open ocean					$8,829.14		$390.60					
pasture	$4,162.64	$4,162.64							$48.70	$1,209.15	$48.71	$1,209.15
redwood 2nd growth					$994.30							
redwood old growth					$1,159.00							
riparian buffer						$10,195.67			$2,699.63	$17,592.96	$435.67	$75,405.06
riparian forest					$10,726.24			$10,632.81				
salt wetland	$30,107.16	$37,922.48			$7,373.68	$19,676.85						
seagrass							$29,456.20					
shrub/scrub									$210.96	$7,098.61	$210.96	$7,098.61
spotted owl habitat					$1,217.56							
swamps/floodplains							$30,349.00					
temperate forest							$468.10					
terrestrial												
tidal marsh							$15,484.50					
tropical forest							$3,110.85					
urban green space	$4,184.60	$10,334.62			$6,838.10	$7,455.32		$6,772.74	$3,746.62	$14,175.83	3746.623	$12,782.96
vineyards					$2,674.24							
woody perennial	$148.84	$148.84										

Appendix 2 – Land Cover Simplification (2012 USD)

Land Cover Type	Troy and Wilson 2006 – Massachusetts low	Troy and Wilson 2006 – Massachusetts high	Troy and Wilson 2006 – Maury Island, WA low	Troy and Wilson 2006 – Maury Island, WA high	Troy and Wilson 2006 – CA Counties	Liu et al. 2010	Costanza et al. 1997	Qenani-Petrela et al. 2007	Batker et al. 2011 low	Batker et al. 2011 high	Schmidt and Batker 2012 low	Schmidt and Batker 2012 high
agriculture					$2,674.24			$2,650.87	$232.17	$1,699.88	$204.94	$1,589.06
conifer forest					$1,001.62			$993.19				
cropland	$4,180.94	$4,180.94				$69.34	$142.60					
disturbed and urban	$0.00	$0.00										
forest	$1,348.10	$2,963.38	$623.42	$3,833.24		$3,867.84	$1,501.95		$1,049.09	$9,632.02	$973.10	$14,544.21
fresh water bodies (lakes/rivers)	$193.98	$46,583.26				$2,306.23	$13,171.90	$8,752.17	$90.24	$60,352.52	$90.24	$9,489.39
freshwater stream			$1,145.58	$1,945.90								
freshwater wetland	$10,338.28	$46,563.74	$40,195.34	$117,235.90	$13,387.06	$26,212.69	$22,916.75	$13,270.64	$1,783.42	$91,381.69	$28,694.77	$154,024.06
grassland/herbaceous			$143.96	$143.96		$36.18	$359.60		$362.28	$1,820.26	$362.01	$1,820.26
hardwood oak woodland					$535.58			$531.40				
mixed forest					$1,007.72			$998.68				
pasture	$4,162.64	$4,162.64							$48.70	$1,209.15	$48.71	$1,209.15
riparian buffer						$10,195.67			$2,699.63	$17,592.96	$435.67	$75,405.06
riparian forest					$10,726.24			$10,632.81				
shrub/scrub									$210.96	$7,098.61	$210.96	$7,098.61
swamps/floodplains							$30,349.00					
temperate forest							$468.10					
urban green space	$4,184.60	$10,334.62			$6,838.10	$7,455.32		$6,772.74	$3,746.62	$14,175.83	3746.623	$12,782.96
woody perennial	$148.84	$148.84										

Land Cover Type	Troy and Wilson 2006 – Massachusetts low	Troy and Wilson 2006 – Massachusetts high	Troy and Wilson 2006 – Maury Island, WA low	Troy and Wilson 2006 – Maury Island, WA high	Troy and Wilson 2006 – CA Counties	Liu et al. 2010	Costanza et al. 1997	Qenani-Petrela et al. 2007	Batker et al. 2011 low	Batker et al. 2011 high	Schmidt and Batker 2012 low	Schmidt and Batker 2012 high
agriculture	$4,180.94	$4,180.94			$2,674.24	$69.34	$142.60	$2,650.87	$232.17	$1,699.88	$204.94	$1,589.06
disturbed and urban	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00	$0.00
forest	$1,348.10	$2,963.38	$623.42	$3,833.24	$848.31	$3,867.84	$1,501.95	$841.09	$1,049.09	$9,632.02	$973.10	$14,544.21
fresh water bodies (lakes/rivers)	$193.98	$46,583.26	$1,145.58	$1,945.90		$2,306.23	$13,171.90	$8,752.17	$90.24	$60,352.52	$90.24	$9,489.39
freshwater wetland	$10,338.28	$46,563.74	$40,195.34	$117,235.90	$13,387.06	$26,212.69	$22,916.75	$13,270.64	$1,783.42	$91,381.69	$28,694.77	$154,024.06
grassland/herbaceous			$143.96	$143.96		$36.18	$359.60		$362.28	$1,820.26	$362.01	$1,820.26
pasture	$4,162.64	$4,162.64							$48.70	$1,209.15	$48.71	$1,209.15
shrub/scrub									$210.96	$7,098.61	$210.96	$7,098.61
urban green space	$4,184.60	$10,334.62			$6,838.10	$7,455.32		$6,772.74	$3,746.62	$14,175.83	3746.623	$12,782.96

Simplifications

Original	Study	Final
Stream	*Troy and Wilson 2006*	Freshwater Body
Cropland	*Troy and Wilson 2006*	Agriculture
Cropland	*Lui et al 2010*	Agriculture
Cropland	*Costanza et al 1997*	Agriculture
Conifer Forest Mixed Forest Hardwood Oak	*Troy and Wilson 2006*	Forest (values averaged)
Conifer Forest Mixed Forest Hardwood Oak	*Qenani-Petrela et al. 2007*	Forest (values averaged)

Appendix 3 – WTP Function Mean Values by Census Tract

Mean Values by Census Tract for WTP Function

Census ID	County	City	Age (Median)[1]	Sex (% male)[1]	Personal Income (Mean non-family)[1]	Household Income (Mean)[1]	Extra Income[2]	*Education[1]	Rural (%)[3]	Connection (%)[4]	Aware of nitrate issues (%)[6]	Uses Treatment[5]	Uses Bottled H_2O[5]	*Years at Zip Code[5]
1	Yakima	Yakima	30.8	58.9	$15,631	$24,459	$8,828	2.5795	0.1	0.9	0.33	0.1586	0.1982	3.348
2	Yakima	Yakima	25.9	66.3	$29,095	$37,108	$8,013	2.3295	0.1	0.9	0.33	0.1586	0.1982	3.348
3	Yakima	Yakima	40.7	45.5	$25,755	$39,938	$14,183	3.3670	0.1	0.9	0.33	0.1586	0.1982	3.348
4	Yakima	Yakima	48.1	48.1	$67,623	$92,233	$24,610	4.2650	0.1	0.9	0.33	0.1586	0.1982	3.348
5	Yakima	Yakima	38.9	45.7	$54,631	$59,757	$5,126	3.4810	0.1	0.9	0.33	0.1586	0.1982	3.348
6	Yakima	Yakima	23.9	49.7	$19,762	$29,059	$9,297	2.5656	0.1	0.9	0.33	0.1586	0.1982	3.348
7	Yakima	Yakima	29.8	49.2	$24,739	$33,808	$9,069	3.1829	0.1	0.9	0.33	0.1586	0.1982	3.348
8	Yakima	Yakima	42.7	48.1	$47,728	$86,449	$38,721	4.2477	0.1	0.9	0.33	0.1586	0.1982	3.348
9.01	Yakima	Yakima	43.1	47.4	$39,644	$67,436	$27,792	3.9125	0.1	0.9	0.33	0.1586	0.1982	3.348
9.02	Yakima	Yakima	33.4	47.3	$76,006	$64,655	-$11,351	3.6716	0.1	0.9	0.33	0.1586	0.1982	3.348
10	Yakima	Yakima	31.6	47.7	$38,289	$56,881	$18,592	4.0151	0.1	0.9	0.33	0.1586	0.1982	3.348
11	Yakima	Yakima	30.6	48.3	$41,840	$56,986	$15,146	3.5265	0.1	0.9	0.33	0.1586	0.1982	3.348
12.01	Yakima	Yakima	27.2	48.3	$28,799	$44,488	$15,689	3.2381	0.1	0.9	0.33	0.1586	0.1982	3.348
12.02	Yakima	Yakima	29.6	49.8	$28,267	$37,332	$9,065	2.8080	0.1	0.9	0.33	0.1586	0.1982	3.348
13	Yakima	Yakima	33.8	50	$22,807	$39,595	$16,788	2.8833	0.1	0.9	0.33	0.1586	0.1982	3.348
14	Yakima	Yakima	32.9	50.4	$27,244	$41,447	$14,203	2.6857	0.1	0.9	0.33	0.1586	0.1982	3.348
15.01	Yakima	Yakima	26.1	54.6	$32,673	$35,449	$2,776	2.4787	0.1	0.9	0.33	0.1586	0.1982	3.348
15.02	Yakima	Yakima	24.9	50.7	$28,915	$31,633	$2,718	2.2185	0.1	0.9	0.33	0.1586	0.1982	3.348
16.01	Yakima	NA	43.3	50.2	$30,126	$80,435	$50,309	3.5455	1	0	0.33	0.1586	0.1982	3.348
16.02	Yakima	NA	43	48.6	$37,339	$67,251	$29,912	3.6787	1	0	0.33	0.1586	0.1982	3.348
17.01	Yakima	NA	40.6	50.5	$37,377	$60,562	$23,185	3.3934	1	0	0.33	0.1586	0.1982	3.348
17.02	Yakima	NA	31	71.1	$38,969	$66,417	$27,448	3.0461	1	0	0.33	0.1586	0.1982	3.348
18	Yakima	NA	32.6	51.4	$47,933	$59,659	$11,726	2.9284	1	0	0.33	0.1586	0.1982	3.348
19.01	Yakima	Grandview	24.2	50.8	$28,572	$36,875	$8,303	2.5891	0.1	0.9	0.33	0.1586	0.1982	3.348
19.02	Yakima	Grandview	27	49.5	$22,439	$43,400	$20,961	2.7189	0.1	0.9	0.33	0.1586	0.1982	3.348
20.01	Yakima	Sunnyside	23.6	51.4	$26,645	$38,080	$11,435	2.4911	0.1	0.9	0.33	0.1586	0.1982	3.348
20.02	Yakima	Sunnyside	27	49.2	$38,015	$54,021	$16,006	2.8687	0.1	0.9	0.33	0.1586	0.1982	3.348
21.01	Yakima	NA	33.5	52.7	$27,709	$52,133	$24,424	2.7441	1	0	0.33	0.1586	0.1982	3.348
21.02	Yakima	NA	25.6	51.1	$23,637	$52,553	$28,916	2.6351	1	0	0.33	0.1586	0.1982	3.348
22	Yakima	NA	32.8	51.2	$51,657	$61,555	$9,898	3.2367	1	0	0.33	0.1586	0.1982	3.348
27.01	Yakima	NA	25.4	50	$28,686	$51,606	$22,920	2.4555	1	0	0.33	0.1586	0.1982	3.348
28.01	Yakima	NA	42.2	50.8	$30,569	$89,710	$59,141	3.9325	1	0	0.33	0.1586	0.1982	3.348
28.02	Yakima	NA	37.4	49.7	$48,121	$83,644	$35,523	3.8439	1	0	0.33	0.1586	0.1982	3.348
29	Yakima	NA	35.9	50.7	$46,086	$84,553	$38,467	3.2168	1	0	0.33	0.1586	0.1982	3.348
34	Yakima	NA	42.3	50.2	$34,524	$72,252	$37,728	4.0001	1	0	0.33	0.1586	0.1982	3.348
9400.01	Yakima	NA	30.3	50.9	$28,967	$51,804	$22,837	3.2654	1	0	0.33	0.1586	0.1982	3.348
9400.02	Yakima	NA	30.8	51.1	$35,628	$48,036	$12,408	2.8568	1	0	0.33	0.1586	0.1982	3.348
9400.03	Yakima	NA	32.3	51.6	$30,244	$59,704	$29,460	3.1331	1	0	0.33	0.1586	0.1982	3.348
9400.04	Yakima	Wapato	25.1	50.5	$17,369	$35,295	$17,926	2.5641	0.1	0.9	0.33	0.1586	0.1982	3.348
9400.05	Yakima	Toppenish	25.4	51.5	$27,375	$35,128	$7,753	2.5323	0.1	0.9	0.33	0.1586	0.1982	3.348
9400.06	Yakima	Toppenish	23.2	50.9	$28,575	$39,905	$11,330	2.2828	0.1	0.9	0.33	0.1586	0.1982	3.348
117	Benton	Prosser	33.5	48.5	$27,538	$53,719	$26,181	3.4893	0.1	0.9	0.33	0.1586	0.1982	3.348
118	Benton	NA	33.8	52.2	$39,892	$66,340	$26,448	3.1604	1	0	0.33	0.1586	0.1982	3.348

[1] US Census 2010
[2] Extra Income = Household Income - Personal Income
[3] For tracts located within designated city boundaries, 10% assumed rural. Outside of city boundaries, assumed 90%.
[4] For tract located within designated city boundaries, 90% assumed connected to municipal system. Outside of city boundaries, assume 0%.
[5] Mean values from Crutchfield et al 1997 for Mid-Columbia Basin. Updated statistics not available.
[6] Assumed 10% increase from 1997 values, due to increased pulic attention on nitrate in the LYRB
** see attached table for value definition*

Appendix 4 - WTP Summary by Census Tract

Census ID	County	City	Total Population[1]	Total Households[1]	WTP (household/month)	TOTAL/month	WTP (household/year)	TOTAL/yr	WTP as % of annual household income	% below poverty line[1]
1	Yakima	Yakima	3095	908	$56.27	$51,095	$675.26	$613,136	2.76%	54.9
2	Yakima	Yakima	5553	1636	$59.63	$97,551	$715.54	$1,170,617	1.93%	36.1
3	Yakima	Yakima	4521	2122	$51.16	$108,569	$613.96	$1,302,823	1.54%	21.5
4	Yakima	Yakima	7423	3100	$51.43	$159,422	$617.12	$1,913,061	0.67%	4.6
5	Yakima	Yakima	5202	2102	$51.09	$107,398	$613.12	$1,288,772	1.03%	13.6
6	Yakima	Yakima	6953	1973	$58.88	$116,172	$706.57	$1,394,062	2.43%	50
7	Yakima	Yakima	7072	2602	$56.96	$148,205	$683.50	$1,778,455	2.02%	25.9
8	Yakima	Yakima	4495	1861	$56.96	$105,995	$683.47	$1,271,937	0.79%	2.8
9.01	Yakima	Yakima	7411	3145	$53.84	$169,329	$646.09	$2,031,943	0.96%	6.8
9.02	Yakima	Yakima	3905	1537	$52.46	$80,634	$629.55	$967,612	0.97%	13.6
10	Yakima	Yakima	5885	2379	$59.54	$141,656	$714.53	$1,699,874	1.26%	9.6
11	Yakima	Yakima	6931	2586	$58.30	$150,764	$699.60	$1,809,168	1.23%	8.9
12.01	Yakima	Yakima	3928	1332	$59.62	$79,410	$715.40	$952,918	1.61%	19.6
12.02	Yakima	Yakima	6318	2101	$56.13	$117,919	$673.50	$1,415,024	1.80%	29.4
13	Yakima	Yakima	2731	945	$55.14	$52,103	$661.63	$625,239	1.67%	23.5
14	Yakima	Yakima	3444	1203	$54.76	$65,874	$657.10	$790,492	1.59%	22.6
15.01	Yakima	Yakima	6991	1916	$57.07	$109,344	$684.83	$1,312,130	1.93%	31.1
15.02	Yakima	Yakima	2804	780	$56.38	$43,979	$676.60	$527,752	2.14%	39.6
16.01	Yakima	NA	2413	877	$71.24	$62,481	$854.92	$749,769	1.06%	14.5
16.02	Yakima	NA	6854	2731	$68.14	$186,091	$817.68	$2,233,097	1.22%	6.1
17.01	Yakima	NA	3083	1094	$67.98	$74,373	$815.80	$892,480	1.35%	23.5
17.02	Yakima	NA	6023	1940	$76.97	$149,322	$923.64	$1,791,860	1.39%	5.7
18	Yakima	NA	7140	2162	$69.72	$150,737	$836.65	$1,808,839	1.40%	17.3
19.01	Yakima	Grandview	3834	101	$58.79	$5,938	$705.47	$71,253	1.91%	31.8
19.02	Yakima	Grandview	6417	1774	$59.36	$105,298	$712.27	$1,263,572	1.64%	31.2
20.01	Yakima	Sunnyside	8401	2181	$59.50	$129,759	$713.94	$1,557,106	1.87%	32.8
20.02	Yakima	Sunnyside	8600	2445	$58.90	$144,019	$706.84	$1,728,233	1.31%	26.9
21.01	Yakima	NA	1950	579	$70.99	$41,104	$851.90	$493,251	1.63%	21.1
21.02	Yakima	NA	7578	1972	$75.89	$149,661	$910.72	$1,795,936	1.73%	30.1
22	Yakima	NA	7745	2501	$70.13	$175,396	$841.56	$2,104,752	1.37%	23.7
27.01	Yakima	NA	3440	842	$74.33	$62,582	$891.91	$750,988	1.73%	25.1
28.01	Yakima	NA	5359	1899	$74.56	$141,587	$894.71	$1,699,046	1.00%	5
28.02	Yakima	NA	7882	2852	$73.08	$208,421	$876.95	$2,501,057	1.05%	12.4
29	Yakima	NA	6570	2165	$72.87	$157,771	$874.48	$1,893,252	1.03%	15.2
34	Yakima	NA	4382	1653	$71.03	$117,417	$852.39	$1,409,009	1.18%	6.8
9400.01	Yakima	NA	6409	1764	$73.81	$130,193	$885.67	$1,562,320	1.71%	29.5
9400.02	Yakima	NA	4430	1256	$70.66	$88,749	$847.92	$1,064,993	1.77%	33.4
9400.03	Yakima	NA	3682	1095	$73.45	$80,432	$881.44	$965,181	1.48%	22
9400.04	Yakima	Wapato	6777	1664	$59.73	$99,384	$716.71	$1,192,603	2.03%	48.4
9400.05	Yakima	Toppenish	4695	1249	$57.94	$72,365	$695.26	$868,375	1.98%	34.7
9400.06	Yakima	Toppenish	4839	1147	$59.05	$67,726	$708.56	$812,717	1.78%	26.3
117	Benton	Prosser	7076	2484	$58.32	$144,875	$699.88	$1,738,498	1.30%	15.4
118	Benton	NA	6546	1976	$72.23	$142,733	$866.80	$1,712,801	1.31%	21.4

[1] *US Census 2010*

Appendix 5 – WTP Coefficients & Variable Definition

WTP Function Parameters	coefficient (α) value
Constant (x)	1.21732
Age	-0.014442
Sex	0.409802
Personal Income	0.000000
Extra Income	0.000004
Education Level	0.067758
Lives in Rural Area	0.251104
Connected to Municipal System	-0.132924
Heard about Nitrate Contamination	0.077777
Uses a Water Treatment System	-0.305657
Uses Bottled Water	0.089862
Years Living at Current Zip Code	0.097674

Bid Value (B)	-0.024085
A = x+ (sum α values * mean values)	
WTP = -1 * (A/B)	

Education Values

1 =< 8th Grade
2 Grades 9-11
3 High School Diploma
4 Some College
5 College Degree
6 Post Graduate Education

Years at Zip Code

1 =< 1 year
2 1-2 years
3 2-5 years
4 => 5 years