LINKING LANDSCAPE INDICATORS TO GROUNDWATER NITRATE CONCENTRATIONS IN A TRANSBOUNDARY AQUFIER

by

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Abstract

Groundwater aquifers provide nearly half the freshwater used in drinking and cooking. However, in the last century, massive transformations of landscapes have produced enduring impacts on natural resources such as groundwater. Excess nitrate contamination of groundwater is a growing health concern, particularly in agricultural regions. Despite its importance, very few studies have quantitatively linked land use land cover (LULC) and groundwater nitrate concentrations. Furthermore, understanding the impacts of LULC on transnational water resources is especially challenging as multi-jurisdictional data disparities and inconsistencies can complicate monitoring efforts.

Here, I developed a suite of innovative long-term monitoring approaches and evaluated their utility in a well-studied transnational aquifer where elevated groundwater nitrate concentrations are of concern. My overall objective was to develop approaches for examining LULC impacts to groundwater via two primary components. First, I used remote sensing to examine two decades of LULC change surrounding 11 groundwater dependent cities. Second, I created more localized landscape indicators and evaluated their correspondence to long-term trends in groundwater nitrate concentrations. I examined two nested spatial extents spanning the US-Canada border including: small cities throughout the Greater Abbotsford-Sumas Aquifer region as well as the confined extent of the Abbotsford-Sumas Aquifer (ASA) proper. I integrated a unique combination of historical photography, transnational satellite imagery, and groundwater monitoring wells spanning four decades. Throughout the larger region, I found that landscape evenness increased over time driven by greater forest losses in Canada and greater losses of agricultural land in the USA. Within the localized ASA, I determined that groundwater nitrate concentrations could be explained using landscape features measured within the vicinity

of wells. Landscape indicators such as the proportional area of berries, raspberry fields undergoing renovations, as well as forage/pasture were particularly useful. I further determined that long-term trends in nitrate were best explained by historical landscape indicators from two decades prior (as opposed to contemporaneous indicators). Very few studies have examined LULC emphasizing transboundary aquifers and even fewer have quantitatively linked groundwater nitrate concentrations to land use practices. Thus, this work demonstrates a valuable, consistent monitoring approach that is transportable to other regions facing similar challenges.

Lay Summary

Aquifers, which are a collection of wet, underground rocks that allow water to pass through them, supply groundwater for drinking and growing food to more than 2 billion people worldwide. What we do on the land surface can have a significant impact on groundwater below. For example, when farmers apply nitrogen-based fertilizers on soils to improve crop growth, any nitrogen not taken up by plants has the potential to runoff into surface waters or leach down into aquifers. This is problematic as drinking water with elevated nitrate levels can be harmful to human health. Monitoring nitrate contamination is challenging as pin-pointing contamination sources is difficult. Monitoring is especially difficult when multiple countries share groundwater resources. This thesis examines the Abbotsford-Sumas Aquifer which straddles the US-Canada border and has ongoing nitrate contamination issues. I develop affordable and transferable approaches for monitoring land use and land cover impacts on groundwater quality. I determined that land use practices from many decades ago likely impacts groundwater nitrate for a long time.

Preface

This dissertation is comprised of three scientific papers of which I am the first author. The project scope was originally defined by myself and my adviser, Sarah Gergel. Long-term groundwater nitrate data were provided by Environment and Climate Change Canada, Agriculture Land Use Inventory (ALUI) data were provided by Metro Vancouver, and historical aerial images were provided by Dr. Hans Schreier. I performed all research, data analyses, interpretation of results, and manuscript preparation. My co-authors provided advice on methodology and editorial changes.

One chapter from this research is currently published:

- Chapter 3:

Gallagher, T. L., Gergel, S. E. 2017. Landscape indicators of groundwater nitrate concentrations: an approach for trans-border aquifer monitoring. *Ecosphere* 8 12 https://doi.org/10.1002/ecs2.2047

One publication from this research is currently under peer review at a journal:

- Chapter 4: Gallagher, TL. Gergel SE, Guttman, M, Schreier H (2018) Historical land cover influences contemporary groundwater nitrate concentrations for decades.

In addition, I wrote Chapter 1 and Chapter 5. My advisor SE Gergel, Lael Parrott, and I co-authored Chapter 2.

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Δ = change

List of Abbreviations

AAFC – Agriculture and Agri Foods Canada

ACI – Annual Crop Inventory

AIC - Akaike information criterion

ALR – Agricultural Land Reserve

ALUI – Agricultural Land Use Inventory

ASA – Abbotsford Sumas Aquifer

BMP – Best Management Practice

CDL – Cropland Data Layer

DOE – Department of Ecology

ECCC – Environment and Climate Change Canada

EFP – Environmental Farm Plan

FAO – Food and Agriculture Organization

GIS – Geographic Information Systems

Greater ASA – Greater Abbotsford Sumas Aquifer region

IGRAC – International Groundwater Resource Assessment Center

LCD – Land Cover Dataset

LULC – Land use and land cover

MK – Mann-Kendall

MMU – Minimum Mapping Unit

MRLCC - Multi Resolution Land Characterization Consortium

N – Nitrogen

NAPL – National Air Photo Library

NASS - National Agricultural Statistics Service

NCDL – National Cropland Data Layer

NO_3^- Nitrate

NO_3 N L^{-1} – Nitrate as nitrogen per liter

NPS – Nonpoint Source

P – Phosphorus

PLAND – Percentage of landscape

SHEI – Shannon's Evenness Index

USDA – United States Department of Agriculture

USEPA – United States Environmental Protection Agency

USGS- United States Geological Survey

WHO – World Health Organization

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To each and every one of my roommates: Thank you for enduring my late night writing sessions and guitar playing and for making our lovely little house on Mackenzie Street a place of retreat. I really hit the roommate jack pot time and time again. Jessica, I'm so lucky to have moved to Vancouver at the same time as you. You helped make our house a home and paved the way for more wonderful roommates even after your departure to California.

To my parents: Wow! It's the end of a chapter; my time in Vancouver. Thank you both for your love and encouragement and understanding through this entire degree. I know I didn't always make it easy, moving to the opposite corner of the continent and all, but I appreciate you both so much for embracing my decision and exploring a new part of the world with me. I simply couldn't have done this without you two. I love y'all!

To my lab mates: Thank you for your support through this. You made going in to the lab a true pleasure. To Karly and Kevin: thanks for covering for me so I could go to Maui! That was brilliant and precisely the dose of sunshine that I needed!!

To Sarah: I can’t express enough how fortunate I feel to have been welcomed into your lab and into your life. I knew from my first meeting with you that I wanted to be your student and I have enjoyed every second of it. You’ve taught me how to be an ecologist, how to be a confident woman in science, and most importantly how to collaborate with others and use my talents to positively contribute to this messy, diverse, and beautiful world. I look forward to many future visits and projects together!

Dedication

"*The heavens belong to the Lord, but he has given the earth to all humanity*." - Psalm 115:16

This research is dedicated to my niece Laina Vogue Britnell. My journey to Vancouver and the start of this project began just a few days after you were born. Watching you grow into the beautiful, thoughtful, and inquisitive child you are, has been a continual reminder of the importance of fostering a sustainable and resilient future.

Chapter 1: Introduction

1.1 Land Use and Land Cover Change is a Force of Global Importance

Over the last century, the global population has more than tripled bringing about the greatest transformation of landscapes in human history (Roser and Ortiz-Ospina 2017). As a result, land use and land cover (LULC) change has become a force of global importance (Foley et al 2005, Vitousek et al 1997). Agricultural conversion of natural lands is acknowledged as one of the greatest anthropogenic impacts to the environment globally (Ramankutty and Foley 1999, Matson et al 1997). In the last three centuries, the total area of cultivated land has increased 466% (Meyer and Turner 1994). Croplands and pastures currently occupy roughly 40% of the global land surface making agriculture one of the largest terrestrial biomes on the planet, rivaling forest cover in extent, and is expected to increase further (Ramankutty and Foley 1999, Asner et al 2004) (Figure 1.1).

Agricultural expansion as well as agricultural intensification (via use of fertilizer, irrigation, and high yield crop varieties) has greatly increased food production over the last 50 years (Naylor 1996). However, despite such improvements in food production, the long-term sustainability and environmental consequences of agricultural systems are of great concern. Altered flux of nutrients - such as nitrogen and phosphorus - are a large part of this dilemma (Howarth et al 2000, Johnson et al 2010).

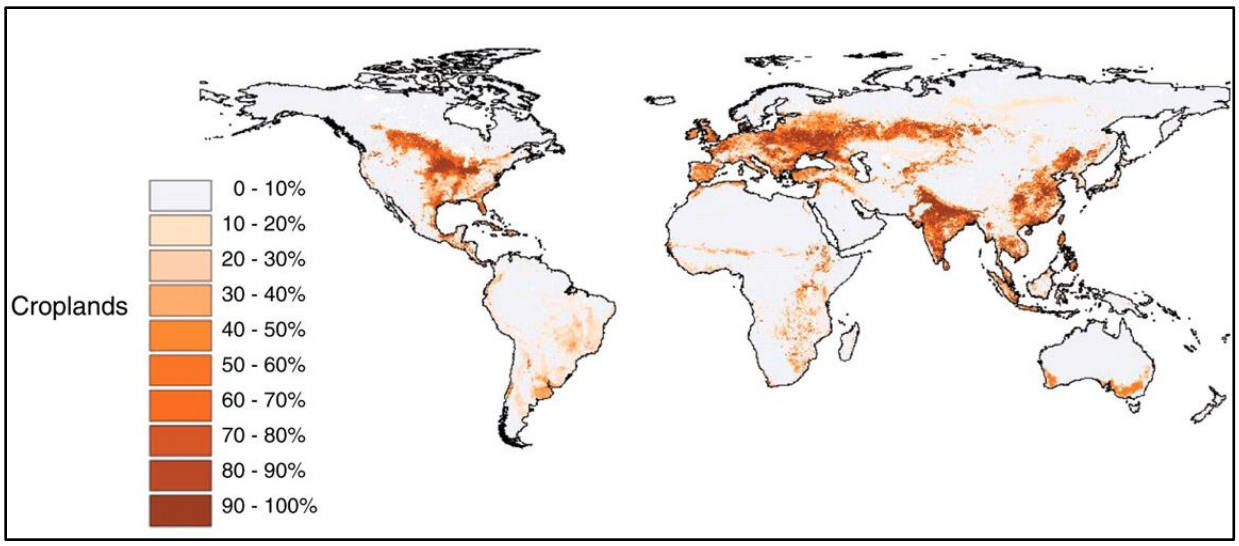

Figure 1.1. The extent of global agricultural land during the 1990s. Source: Jonathan A. Foley et al, Science 2005;309:570-574

1.2 Nitrogen is a solution. It should not be a problem.

The creation of reactive nitrogen (N) via the Haber-Bosch process in the early 1900s prompted the Green Revolution. Widely considered by scientists and historians as the most important invention in modern history, Haber-Bosch brought about the ability to manufacture fertilizer using the atmosphere's abundant nitrogen reserves, resulting in large increases in crop yields to support the planet's growing population (Smil, 2001) (Figure 1.2). While the Haber-Bosch process has helped increase global food production, the increase in global reactive nitrogen has also taken an unforeseen toll on ecosystems and has fundamentally impacted the way humans practice agriculture (Sebilo et al 2013, Kaushal et al 2011, Fields 2004).

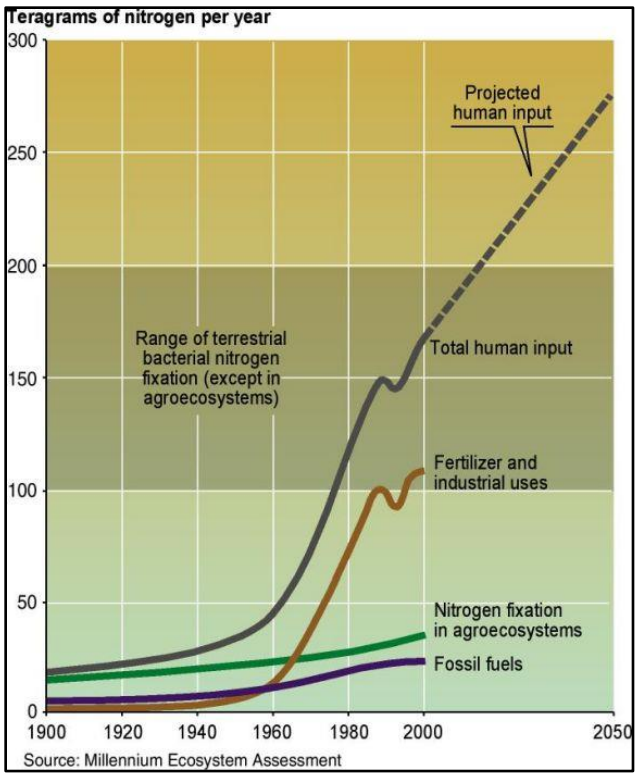

Figure 1.2. Since the creation of the Haber - Bosch process, synthetic nitrogen use has seen a four-fold increase (Millennium Ecosystem Assessment, 2005).

1.3 The global food supply is dependent on anthropogenic N

Presently, synthetic N fertilizers supply over half of the need of the world's crops (Smil 2011). Since the invention of synthetic fertilizers, the number of people supported per hectare of arable land increased from 1.9 persons in 1908 to 4.3 persons in 2008 (Erisman et al 2008). Without synthetic N, today's soils would not be able to grow the amount of food needed to support global dietary demands. Together with new high yielding, short-stalked varieties and chemical protection, yields of wheat and rice worldwide tripled and quadrupled during the 20th century opening the door for farms to move away from the millennia-old system of cycling and re-cycling nutrients and organic matter in each farm (Smil 2011).

Prior to Haber-Bosch, diversified farms often produced food for humans as well as livestock. The resulting manure from cattle and poultry, etc., was then applied back onto the land. Synthetic nitrogen helped bring about the age of industrial agriculture whereby 'modern' farming of homogenous monocultures has replaced diversified, multi-dimensional farms in many regions (Reganold et al 2005). This disconnect between animal husbandry and row crop agriculture disrupted previously more localized flux and re-use of nitrogen, creating a two-fold problem: 1) the need to import synthetic fertilizer onto farms and 2) the necessity to dispose of excess animal manure (Sharpley at al 1994, Follett 2001), often at very disjoint locations. Globally, livestock has increased in the last 50 years with the number of cattle increasing from 942 million in 1961 to 1.4 billion in 2016. Production of pork and poultry has also increased from 406 million and 2.9 billion animals in 1961 to 981 million and 22.7 billion animals in 2016, respectively (FAO, 2018). Once a highly valued resource, animal manure has become a waste disposal problem (Pollan 2006, Montgomery 2007, Hager 2008). This increase in application of industrial N along with the surplus of manure N has helped create several ecological problems including surface water eutrophication and groundwater contamination (Rabalais 2002, Wu and Sun 2016).

1.4 Contamination of groundwater is a global concern

Increasing use of synthetic and organic fertilizer has contributed to a reduction in N use efficiency and increased the potential for nitrate leaching into groundwater over the last 30 years (Townsend and Howarth 2010). Globally, groundwater provides approximately 45% of freshwater used for drinking and cooking, and an additional 24% of water used in irrigated agriculture (Van der Gun 2012). As global landscapes transition to more intensive agriculture and urbanization, groundwater is increasingly susceptible as land use and land cover (LULC)

change can have significant repercussion on recharge rates and quality (Eckhardt and Stackleberg 1995, Loague and Corwin 1998, Kolpin 1997). As a result, nitrate (NO_3^-) contamination of groundwater is a global concern (Goodchild 1998, Joosten et al 1998, Birkinshaw and Ewen 2000, Saãdi and Maslouhi 2003, Kyllmar et al 2005, Liu et al 2005, Almasri 2007) (Figure 1.3).

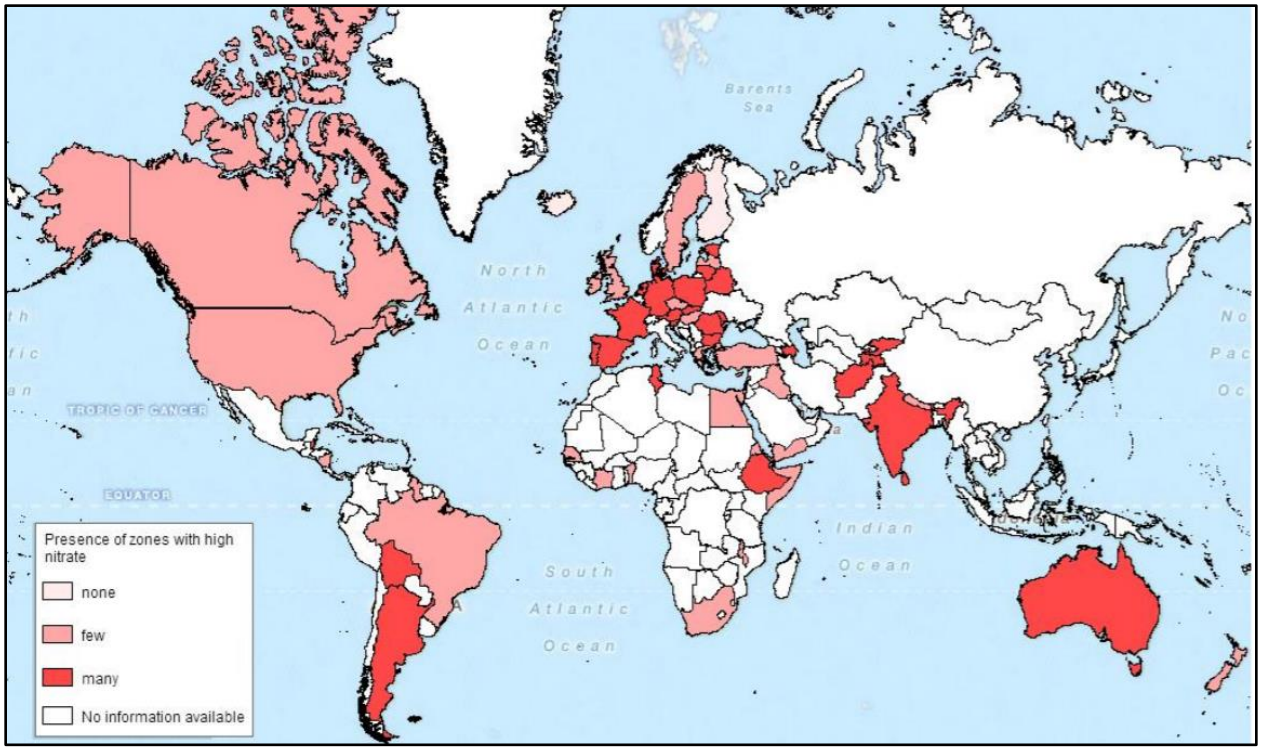

Figure 1.3. Countries with zones of high groundwater nitrate concentrations (IGRAC, 2012). The map shows aggregated data per country, classified from none, few and many zones where high concentration of nitrate have been reported. Based on a literature review, the map demonstrates the percentage of regions with high nitrate contamination in the world. One drawback of this map is that the legend is of qualitative range, without specifying quantitative definition for "high nitrate", "many" and "few."

In response, the World Health Organization (WHO) has established a maximum threshold of 10mg/L NO_3^--N for drinking water aimed to avoid problems such as hypoxemia in infants (Canadian Council of Ministers of the Environment 2014; United States Environmental

Protection Agency 2017). Other potential health effects of excess nitrate in drinking water include reproductive problems and high risk of non-Hodgkin's lymphoma (Weyer et al 2001, Weisenburger 1991, Ward et al 1996). Adverse effects may also be possible below WHO guidelines, however, as long-term exposure to nitrate$^-$ in community water supplies as low as 2 - 4 mg NO_3^- - N L^{-1} has shown possible links to bladder and ovarian cancer (Weyer et al 2001).

1.5 Landscape indicators are an innovative long-term approach to groundwater monitoring

In order to find solutions to groundwater quality issues, we must first explore the context of and potential sources causing the contamination. Land use patterns, including historical, current, and future anticipated changes in land use, are part of this challenge. The impacts of LULC change on atmospheric components of the hydrologic cycle (regional and global climate) are well-recognized (Bonan, 1997; Pielke et al 1998; Pitman et al., 2004). For example, land cover changes have been linked to altering albedo, increasing regional temperatures and reducing precipitation (Pitman et al 2004, Cao 2015). Though the impacts of LULC on atmospheric components of the hydrologic have been explored, fewer studies examine the impacts of LULC change on subsurface components of the hydrologic cycle (Scanlon et al 2005). These impacts are multifaceted, linked not only to agricultural expansion but also to the loss of agriculture to urban expansion, both of which impact nitrogen flux to surface and groundwater.

One approach to understanding how LULC influences groundwater is through landscape indicators which can be easily created using GIS and remote sensing. Landscape indicators quantify the amount and arrangement of land cover (such as percent agriculture and percent forest cover) and characterize the physical structure of vegetation on the land surface (Meyer and

Turner 1994*)*. They allow for an affordable, broad-brush approach to characterizing the landscape and identifying potential LULC impacts. A long-standing, well-developed body of research examines the correlations between landscape indicators and surficial aquatic ecosystems (Gergel et al 2002, Allan 2004, Johnson and Host 2010). As such, the potential mechanisms shaping the correlations between LULC and water quality are well understood in a qualitative sense. For example, the amount of agriculture in a basin may be associated with higher stream sediment and/or nutrients concentrations (Blake et al 2012, Arheimer and Liden 2000, Osborne and Kovacic 1993). Nonetheless, the strength of quantitative predictions can vary greatly depending upon the landscape indicator used and the region over which it is applied.

While a plethora of research has examined landscape indicators relative to surface waters (Hale et al 2004, Mallin et al 2000), few have examined landscape indicators within the context of groundwater aquifer monitoring (Gurdak and Qi 2006, Keeler and Polasky 2014), despite the links between surface and groundwater systems. Furthermore, much of the landscape indicators research is highly correlational - using only *land cover-based* indicators - and is generally lacking in a deeper exploration of mechanisms associated with *land use practices*. To help fill these knowledge gaps, I seek to identify fine-scale landscape indicators that better account for land use practices likely related to nitrate loading. My work helps improve the development of landscape indicators for groundwater by identifying and better incorporating potential mechanisms and processes acting above and below the land surface (Sophocleous 2002).

1.6 Research approaches and objectives

Given the need for new spatial and temporal approaches for groundwater quality monitoring, in this thesis I develop and explore multiple innovative tools useful across an extended timeframe and across large regions. My overall objective is to develop approaches for

examining LULC impacts to groundwater systems. I accomplish this in two main parts: 1) examining 2 decades of change in LULC pattern along the urban-rural gradient for 11 groundwater dependent cities and 2) creating and linking landscape indicators to long-term trends in nitrate concentrations in a well-studied aquifer. I explore the region surrounding the Abbotsford-Sumas Aquifer (ASA) which straddles the US and Canada border (Figure 1.4). The ASA, which spans Northwestern Washington state and Southwestern British Columbia, has a long history of persistent elevated nitrate concentrations. The many complexities of this problem have challenged managers, farmers, and policy makers in both countries who have initiated a wide variety of nutrient management strategies - with little apparent success - in reducing $nitrate^-$ concentrations. In my work, I address two nested scales: the Greater Abbotsford-Sumas Aquifer (Greater ASA) region, as well as the localized extent of the ASA in particular (Figure 1.4).

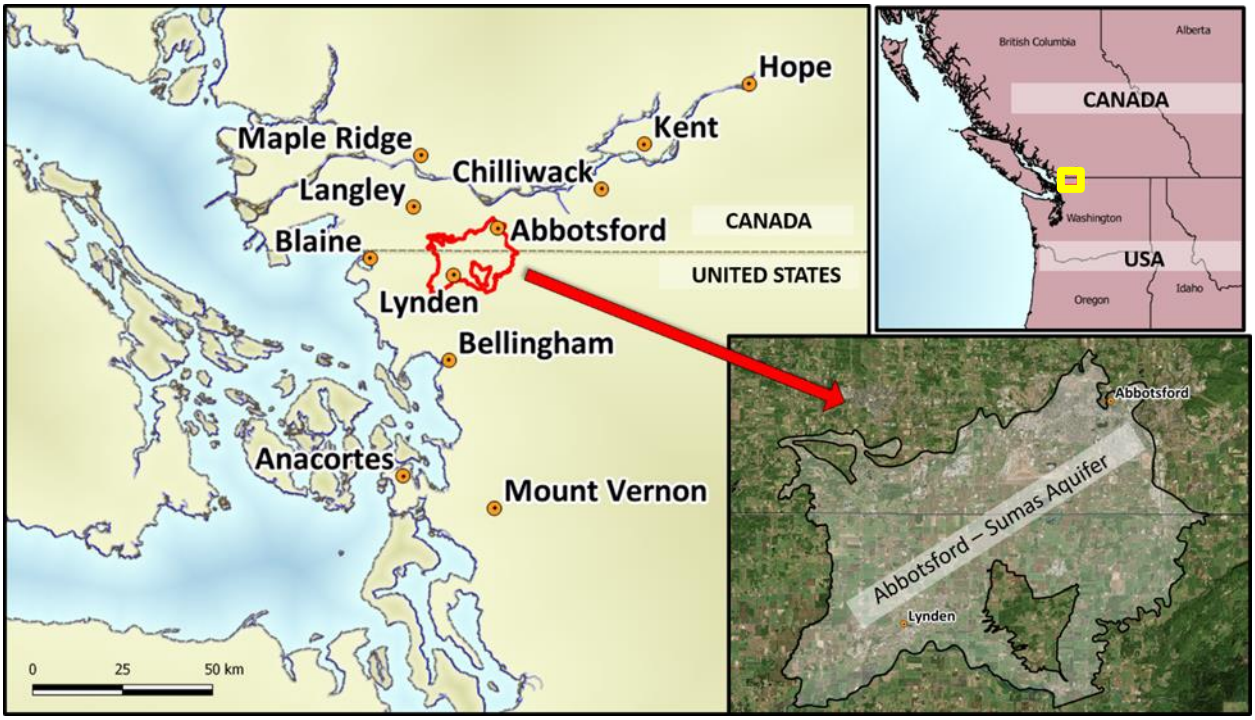

Figure 1.4. The Abbotsford-Sumas Aquifer (ASA) is a shallow unconfined aquifer straddling the US-Canada border. Elevated nitrate concentrations have plagued the aquifer in recent decades. This thesis sets out to study linkages between LULC and groundwater using landscape ecological approaches over two spatial extents: the localized ASA area (delimited in red) as well as 11 US and Canadian cities (shown as yellow points) throughout the Greater ASA region.

The Greater ASA region is an ideal study location for several reasons. First, many of its cities and municipalities are heavily dependent on aquifers as their primary drinking source. Second, both agricultural expansion and urbanization are occurring simultaneously throughout the Greater ASA and these LULC changes are transforming the social-ecological landscape and impacting linked surface-groundwater systems. Thirdly, these trans-boundary aquifers are influenced by policies and economic drivers of two countries. Thus, my cross-border approach enables comparative research relevant to both countries. Through such comparisons, I determine how landscape patterns differ between the two countries and determine whether past (and

potential future) LULC impacts on groundwater might also differ. Lastly, the ASA in particular has a consistent, very well-developed long-term groundwater monitoring program, along with an abundance of historical imagery. Integrating this extensive and somewhat unique suite of long-term data can help shed new light on groundwater issues in this particular region yet also provide a valuable adaptable approach for monitoring in other cross-border aquifers where LULC is of concern.

This thesis is comprised of three research components linking groundwater and land cover change. To provide broad context for landscape changes occurring throughout the region, in Chapter 2 I explore where, and at what rate, land conversions are taking place via a trans-border regional LULC change analysis of the Greater ASA region. In Chapters 3 and 4, I narrow this broader perspective to focus directly on how LULC affects groundwater nitrate concentrations in the ASA specifically. Next, I discuss the overall goals and original contributions of each chapter.

Chapter 2: Trans-border monitoring of landscape pattern

Monitoring transboundary aquifer bodies is complicated by the complexities associated with multi-jurisdictional governance, disparities in data collection, and inconsistencies in geospatial data among countries. To broaden our understanding of the spatiotemporal landscape footprint of agricultural expansion and intensification, which is often followed by urbanization, I quantify changes in LULC patterns for cities where groundwater is an important source of drinking water. Using a myriad of geospatial data sources and landscape analysis, I seek to answer: 1) How do landscape patterns change over time, and throughout the urban-rural gradient? 2) Do these patterns differ between US and Canadian cities? To answer this, I examine 11 municipalities and small cities (<150,000 residents) reliant on groundwater for drinking water

within the Greater ASA region (Figure 1.4). I identify how landscape patterns change over time along the urban-rural gradient of these cities as they expand into the suburban fringe and surrounding agricultural areas. The over-arching theme of this chapter is to quantify spatial and temporal trends in landscape patterns and to discuss their potential implications for groundwater water quality. This work also helps provides broad regional context for Chapters 3 and 4.

Chapter 3: Are there linkages between contemporary nitrate concentrations and contemporary land cover?

After characterizing landscape change throughout the broader region, next, I narrow in on how LULC impacts nitrate concentrations in a specific aquifer for which detailed monitoring data exist. I ask two questions: 1) Are there temporal trends in nitrate concentrations over time? and 2) How well do landscape indicators help explain patterns of groundwater nitrate concentrations? To accomplish this, I first test for statistical trends in nitrate concentrations over time. Second, I combine cross-border geospatial data to develop landscape indicators characterizing likely N sources and examine their correspondence with groundwater nitrate concentrations. Through linking these contemporary landscape indicators to contemporary nitrate concentrations, I aim to help portray the current state of the aquifer. This work also provides an important transportable approach that is highly relevant to other regions facing similar management challenges and lays the foundation for understanding how to quantify potential sources of nitrogen using landscape indicators.

Additionally, I seek to answer three subsidiary objectives, to determine: a) the spatial scale (or distance) over which landscape indicators should be measured; b) whether incorporating groundwater flow direction into indicators improves model results; and c) whether

effective landscape indicators of nitrate concentrations in the USA and Canada differ. To do so, I calculate landscape indicators within terrestrial zones of influence (composed of differently sized radii) surrounding monitoring wells and account for directionality of subsurface flow. Finally, I compare my results for the US and Canada. I hypothesize that incorporating distance and directional flow into landscape indicators will increase their predictive power and thus improve our understanding of the factors contributing to high nitrate concentrations throughout the ASA. While this chapter examines recent nitrate concentrations from 2005-2013 and links it to 2012 land cover, in the next chapter, I look even deeper in time to link historical (1996-2015) nitrate concentrations with historical land cover.

Chapter 4: Are there linkages between historical nitrate and historical land cover?

Historical LULC can influence water chemistry for decades. Despite its importance, very few studies have quantitatively evaluated historic land use patterns and their correspondence with groundwater nitrate concentrations. Building off the previous chapter, I evaluate even longer-term trends to assess potential lagged effects of LULC on nitrate concentrations. I use high spatial resolution imagery to map fine-scale features (such as field hedgerows) and land use practices (such as replanting of raspberry fields) potentially linked to nitrate loading. I also incorporate groundwater flow direction to better capture areas potentially contributing to nitrate leaching.

Extending my analysis of the ASA aquifer by an additional 30 years, I ask two primary questions regarding landscape legacies: 1) Are long-term groundwater nitrate concentrations changing over time and space? 2) What is the relative importance of historical versus contemporary LULC in explaining groundwater nitrate concentrations? To accomplish this, I

examine trends in groundwater nitrate concentrations at monitoring wells across the aquifer in a similar fashion to Chapter 2, but I add new information from landscape indicators originally mapped on historical aerial imagery from the 1970s. I hypothesize that correlations between present day nitrate concentrations and historic LULC (1974 and 1996) may be important because of lagged effects of land use practices. Historical imagery can help us understand persistent "legacy" effects of land use on groundwater, an essential part of creating effective policies and management strategies for improving water quality.

Chapter 5 (Conclusion): A look towards the future of transboundary aquifer management

In my concluding chapter, I provide a summary of my key findings, discuss limitations of my approach and its significance, both locally and globally, and I explore directions for future research and community-based management. I also posit that there are potential benefits of taking a more socio-ecological approach to groundwater monitoring whereby techniques such as future scenario planning can garner greater community involvement in decreasing nitrate concentrations over time. This synthetic and integrative research has global implications as management of oft-shared trans-border groundwater resources can result in trans-national conflicts.

Chapter 2: Transboundary Monitoring of Landscape Pattern

2.1 INTRODUCTION

In recent decades, agricultural transitions have come under increasing scrutiny. Along with transitions to more intensive agriculture, agricultural lands are being lost to urban land uses with accompanying increases in impervious surfaces across the landscape. Such landscape transitions have become a worldwide concern (Alexandratos and Bruinsma 2012) as population growth pushes cities to expand their boundaries. This urban expansion can impact previously disjointed landscapes where high density populations are concentrated in one location yet agricultural and wildlands are dispersed elsewhere. Patterns of land conversion in outlying exurban environments are of increasing as these changes impact ecosystem services, biodiversity, and aquatic systems (Johnson 2001, Grimm et al 2008, Han et al 2015).

Land cover change surrounding smaller urban centers is particularly important as they are key locations undergoing agricultural transitions. Land conversion across broader scales (metropolitan, state, or nation-wide scales) has received great attention, yet less attention has been paid to smaller cities (Goldewijk 2001, Foley et al 2005, Keys et al 2007). Half of the US population live in rural areas or small jurisdictions under 25,000 people (such as towns, boroughs, villages and townships) (Cox 2008). Such rural communities can be highly dependent on groundwater for drinking water supplies. Groundwater is particularly susceptible to LULC change as increases in impervious surfaces and loss of wetlands can have significant repercussion on recharge rates and water quality (Loague and Corwin 1998, Kolpin et al 2002).

Of increasing interest to ecologists, and of longstanding interest to regional planners, is the rural-urban gradient (McDonnell and Hahs 2008, Ramachandra et al 2015) which emphasizes differences along a spatial continuum from urban to rural areas (Haase and Nuissl 2010). As

defined, this gradient primarily extends from the urban core to rural outskirts (Kroll and Kabisch 2012). Urban to rural gradients have been widely used to analyze impacts of landscape change on non-point source (NPS) pollution (Lovett et al, 2000, Gingrich and Diamond 2001), water quality (Wear et al, 1998), ecosystem services, avian diversity and richness (Blair 1996, Mortberg 2001), and invertebrate communities (Walsh 2006). Despite the many studies quantifying changes in landscape patterns along urban-rural gradients, few have examined smaller cities. Ecological studies of urban-rural gradients remain focused on larger metropolitan areas (McDonnell and Picket 1990, Luck and Wu 2000).

In this chapter, I am interested in understanding LULC patterns in smaller cities reliant on aquifers for drinking water. Many of the landscapes surrounding such cities are undergoing rapid agricultural transitions. Using an approach which is designed to be transportable across jurisdictions, I quantify trends in landscape patterns from 1990 to 2015 surrounding 11 urban centers. I ask the following questions: *How do landscape patterns change over time, and throughout the urban-rural gradient? Do these patterns differ between US and Canadian cities?* While very few studies have examined spatio-temporal landscape patterns of smaller urban areas, even fewer have compared cities across international borders (Desender et al 2005, Clergeau et al 1998). Studying smaller cities will yield a richer picture of urban form and function and potential impacts on groundwater (Bell and Jayne 2009). Further, this general approach can be used to assess long-term landscape changes in a variety of different agricultural socio-ecological landscapes.

2.2 METHODS

2.2.1 Study Area

The Greater ASA region is a highly agriculturally productive region in the central Fraser Valley spanning the section of US-Canada border that stretches from southwest British Columbia into Whatcom County, Washington (Figure 2.1). Several international waterbodies in this region include Boundary Bay, the Nooksack River Watershed, and the Abbotsford-Sumas Aquifer (ASA). Within the Greater ASA region, the dominant crop consists of red raspberry with significant forage grass and pasture (Zebarth, 2015). There is also a substantial amount of poultry and dairy operations. From 2006 – 2016, the number of chickens and hens within the Canadian portion of the Fraser Valley increased from 11.3 to 13.7 million while the number of cattle and calves increased from just under 93,000 to 103,000 (Ministry of Agriculture 2016). Within this same area, the human population increased by 18,000 and total farmland area grew from just over 56,500 ha to nearly 61,000 ha from 2001-2016. Within the USA portion of the Fraser Valley (Whatcom County), the population increased by 16,000 people from 2005-2014 whereas total farmland increased nearly 13% from 41,500 hectares (2007) to 46,800 hectares (USDA, 2018). Within this region, I chose cities dependent on groundwater as a drinking water source, including: the Canadian cities of Abbotsford, Chilliwack, Hope, Kent, Langley, and Maple Ridge, and the US cities of Anacortes, Bellingham, Blaine, Lynden, and Mount Vernon. Table 2.1 indicates population and groundwater resources of each city.

Figure 2.1 Map of Greater Abbotsford-Sumas aquifer area. For this chapter I examined changes in LULC patterns in 11 cities in the US and Canada including: Washington cities of Anacortes, Blaine, Bellingham, Lynden, and Mount Vernon, and British Columbian cities of Abbotsford, Chilliwack, Hope, Kent, Langley, and Maple Ridge.

Table 2.1 Eleven groundwater reliant cities in the US and Canada with populations <150,000. Each city obtains at least some portion of their drinking water from regional aquifers, as noted. Source: US Census, 2018; Stats Canada 2018.

Location	City	1990 Population	2015 Population	Drinking Water Source
CANADA	Abbotsford	85,932	141,397	Abbotsford-Sumas aquifer
	Chilliwack	54,152	83,788	Sardis-Vedder aquifer
	Hope	7,837	6,181	Kawkawa Lake aquifer
	Kent	4,322	6,067	Agassiz aauifer
	Langley	85,810	117,285	Abbotsford-Sumas aquifer
USA	Anacortes	11,628	16,387	Whidbey Island aquifer
	Bellingham	53,850	87,574	Abbotsford-Sumas aquifer
	Blaine	2,726	5,164	Blaine-Sumas aquifer
	Lynden	5,979	13,517	Abbotsford-Sumas aquifer
	Mount Vernon	18,496	34,590	Skagit Delta Surficial aquifer

2.2.2 Land use data and classification

Surrounding these cities, I examined land cover patterns and their change over time. To do so, I first assembled and harmonized geospatial data from circa 1990 and 2015 for both countries. In Canada, I used the Agriculture and Agri Foods (AAFC) Land Cover Dataset (LCD) (1990) and AAFC Annual Crop Inventory (ACI) (2015). The 1990 AAFC-LCD contains 15 unique LULC categories whereas the AAFC-CID contains 64 categories. Unable to obtain free 1990 land cover data for the US-portion of my study site, I obtained data for the year 1992 via the Multi Resolution Land Characterization Consortium (MRLCC) National Land Cover Database (NLCD). I acquired 2015 land cover from the United States Department of Agriculture (USDA) Cropland Data Layer (CDL). The NLCD contains 21 categories and the USDA-CDL contains 84 classes. Characteristics of these datasets are further explained in Table 2.2.

To harmonize these datasets to support multi-year and cross-border comparisons, I reclassified land cover into eight categories: Cropland, Forest, Urban, Vegetation (land that is not forest, wetland or crop), Water, Wetland, Other Land (all other land types not mentioned), and Unclassified. Reducing the total number of classes by grouping similar land cover types improves class-level accuracy (Aronoff 1982, Olofsson 2014). For this process I manually examined every class within each data set and grouped each into the most appropriate higher-order land cover grouping. This process was exceptionally time consuming requiring careful iterative verification of my decisions and clarification of the many detailed assumptions and technical decisions underlying these original classifications from several jurisdictions. As the overall accuracy for each data set used was over 80% and class accuracy for classes such as different forest types, berry types, and developed lands were all 80-90%, 70-96%%, and 75-85%, respectively (Table 2.2). Thus, my simplified groupings of class types are at least as accurate as

the classes in these original data sets, with potential for increased class-level accuracy. A more detailed description of land cover types is given in Table 2.3. For ease of reference in the remainder of this chapter, I refer to 1990 Canadian and 1992 US land cover as 1990s LULC.

Table 2.2. Description of geospatial datasets used for landscape pattern analyses. Land cover datasets all had a 30m resolution and had between 15-84 classes. Datasets were manually aggregated and reclassified into 8 classes for consistency.

Data Set	Source	Spatial Resolution	# of Classes	Overall Accuracy (%)	Individual Class Accuracy (%)		Extent of Coverage	Year
Annual Cropland Inventory (ACI)	Agriculture and Agrifoods Canada (AAFC)	30 m	64	86.27	Forest	90.7	Canada	2015
					Blueberry	86.5		
					Urban	84.3		
Cropland Data Layer	United States Department of Agriculture (USDA)	30 m	84	90.9 %	Mixed Forest	82.7	USA	2015
					Caneberries	96.4		
					Developed High Density	79		
Land Cover Dataset	Agriculture and Agrifoods Canada (AAFC)	30m	15	84	Forest	91.8	Canada	1990
					Cropland	71.5		
					Urban	87.6		
National Land Cover Dataset (NLCD)	Multi Resolution Land Characteristics Consortium (MRLCC)	30m	21	80	Forest	80.5	USA	1992
					Caneberries	83.4		
					Developed	76.4		

Table 2.3. Explanation of land cover classes used in landscape pattern analysis. Original data sources (from Table 2.2) were harmonized into these eight classes for comparative purposes.

Land Use Class	Land Cover Class Descriptions (2015)
Cropland (Cr)	Greenhouses, pasture/forage areas, row crops, and other crops
Urban/Built (Ur)	Predominantly built-up or developed land such as urban areas, roads, railways, buildings, pavement, industrial sites, mine structures, etc., as well as associated vegetation.
Vegetation (Veg)	Any vegetation not included in cropland, forest, and wetland classes. Predominantly woody vegetation of relatively low height (generally +/-2 meters). May include grass or some wetlands with woody vegetation, regenerating forest. Predominantly native grasses and other herbaceous vegetation. This category may also include some shrub land cover.
Forest (Fo)	Coniferous, broadleaf, mixed wood
Other Land (Ot)	Predominately non-vegetated and non-developed. Includes: glacier, rock, sediments, burned areas, rubble, mines, and other naturally occurring non-vegetated surfaces. Excludes agricultural fallows.
Wetland (We)	Water table near/at/above soil surface long enough to promote wetland or aquatic processes (semi-permanent or permanent wetland vegetation, including fens, bogs, swamps, sloughs, marshes, etc.).
Water (Wa)	Water bodies (lakes, reservoirs, rivers, streams, salt water, etc.).
Unclassified (Un)	Background of raster images

2.2.3 Characterizing the Urban-Rural Gradient

There are two well-established approaches for examining urban-rural gradients. One approach uses a series of concentric rings originating in the city center expanding outward to rural outskirts (Kroll 2012). A second approach uses directional linear transects originating in the city center extending into rural areas (Hah and McDonnell 2006, Zhou and Wang 2011).

The concept of concentric rings around city centers was first developed as a model of residential differentiation (i.e. socio-economic and racial differences between city and suburb areas). More recently, the approach has been applied to map ecosystem services, fragmentation, and landscape patterns and changes (Burgess 1967, Schneider and Woodcock 2008, Solon,

2009). The concentric ring approach assumes cities exhibit similar spatial patterns in all directions; however cities are not necessarily isotropic (Zheng, 1991). Cities with monocentric urban expansion (as opposed to cities with multiple nuclei) are most suitable for examination using the concentric ring approach. Urban areas in North America typically have a densely populated urban core surrounded by rings of diminishing landscape modification (Dickinson 1966, Forman and Gordon 1986). Nearly all cities in this study exhibited one major urban nucleus and thus reflect this basic assumptions behind the concentric ring approach.

Furthermore, the geography of most of the cities in this study permits equal expansion in all directions (as opposed to other places where expansion is limited in one or more direction by coastlines, mountains, or other geographical features). Where expansion is restricted, the transect approach to examining the urban-rural gradient is more suitable so that patterns can be characterized based on direction (Luck and Wu 2002, Banzhaf et al 2009). Lastly, the transect approach would hinder an important objective of this work - to compare cities north and south of the border - as some transects would proceed into the adjacent country.

Thus, for many reasons, the concentric ring approach was deemed most suitable for this research several. As such, I used the concentric ring gradient method to spatially and quantitatively describe changes in landscape patterns over time. Surrounding each city, I used ESRI ArcMap 10.5 (2015) to delineate a series of concentric rings surrounding the city center (Figure 2.2) which was defined as the geographical centroid of urban/built-up areas. I used a maximum radius of 10 km as a compromise between the largest possible size that also minimized the influence of urban sprawl radiating from neighboring cities. More specifically, for larger-sized cities such as Abbotsford and Bellingham, a minimum 10-km radii distance was

necessary to encircle rural areas; whereas for smaller cities (i.e. Lynden), interference of sprawl from neighboring cities was a concern beyond a 10 km radii.

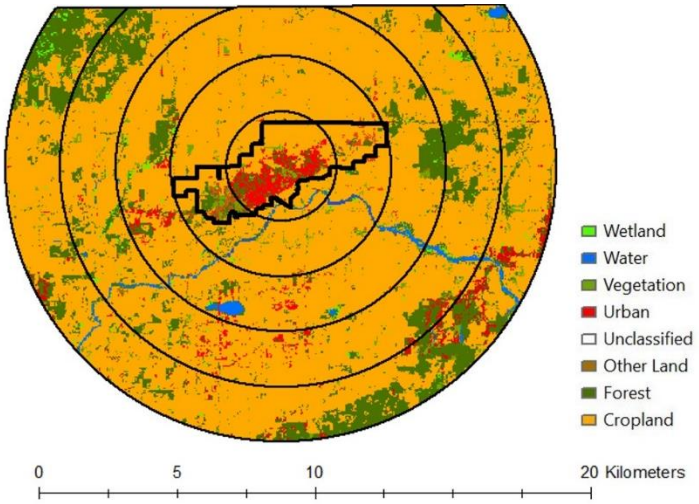

Figure 2.2. Example LULC analysis along the urban-rural gradient for the city of Lynden, WA, USA. Using harmonized land use and land cover data (explained in Table 2.3), LULC was clipped within concentric rings representing distance categories along the urban-rural gradient.

2.2.4 Spatial Pattern Analysis

I examined trends in landscape patterns over time as well across the urban-rural gradient. To do so, I selected landscape pattern metrics based on their ecological significance as well as those most commonly used in land conversion studies in order to bolster their comparative relevance. I examined class-level pattern metrics for three distinct land cover classes (urban, agriculture, and forest) as well as some landscape-level pattern metrics characterizing all classes simultaneously. Metrics included percentage of landscape (PLAND) for each class individually, as well as Shannon's evenness index (SHEI) which included all eight classes in its calculation. I used the FRAGSTATS spatial pattern analysis program ver. 2.0 (McGarigal and Marks 1995). Algorithms used in these calculations are listed in Appendix C of the 171 FRAGSTATS manual (McGarigal and Marks 1995).

PLAND equals the percentage the landscape comprised of the corresponding patch type. Like total class area, it is a measure of landscape composition important in many ecological applications. PLAND is a relative (proportional) measure. Thus it is a more appropriate measure of landscape composition than class area for comparisons involving landscapes of different sizes (McGarigal and Marks 1995), and in this case, comparisons among concentric rings with different total area.

SHEI is a measure of the evenness of classes, expressed such that an even distribution of area among classes results in maximum evenness. Shannon's evenness index ranges from 0-1. SHEI = 1 when the area occupied by all classes is perfectly even (i.e. equal proportional abundance). In contrast, SHEI = 0 when the landscape contains only one class (i.e. no diversity) and approaches 0 as the distribution of area among different classes becomes increasingly uneven (i.e. increasingly dominated by one or few classes).

Statistical Analysis

I calculated change in percentage of landscape (PLAND) between two time periods (1990 and 2015) within the entire 10 km radius of each city, as well as compared SHEI among five distance intervals along the rural-urban gradient (0-2 km, 2-4 km, 4-6 km, 6-8 km, 8-10 km). Using these metrics, I also compared cities in the USA and Canada. To evaluate trends along the urban-rural gradient, I conducted a Mann-Kendall trend test of PLAND (for urban, agriculture, and forest classes separately) across all five distance classes. To do this, I first plotted the spatial trend for each of the three land cover classes for each city for 1990 and 2015. I then conducted the Mann-Kendall test to determine the p-value, Sen's Slope, and Kendall statistic. I then compared 1990 and 2015 results to determine how land cover trend changed over time for each city.

2.3 RESULTS

2.3.1 For most cities, urban area increase at the expense of agricultural and forest land

Percentages of different land cover classes for each city in 1990 and 2015 are presented in Figure 2.3. Changes in land cover along the urban-rural gradient over this time period for all cities are plotted in Figure 2.4. Because the greatest percentage changes in land cover were seen for urban, agriculture, and forest classes, these results focus on these three land cover classes.

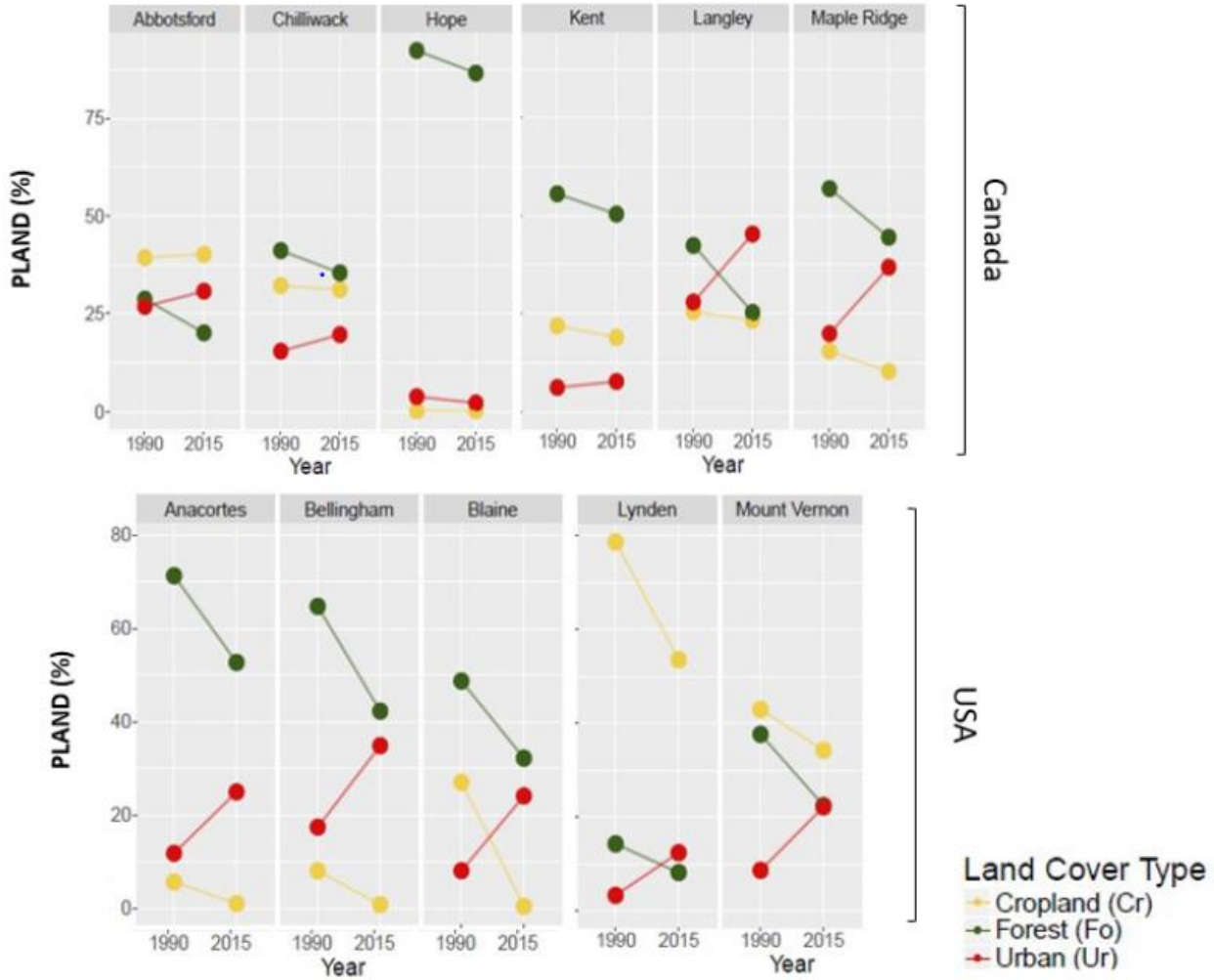

Figure 2.3. Percentage of land cover types for each city in 1990 and 2015. This represents land cover within 10 km radius of the city center. Additional information can be found in Appendix Table 1.

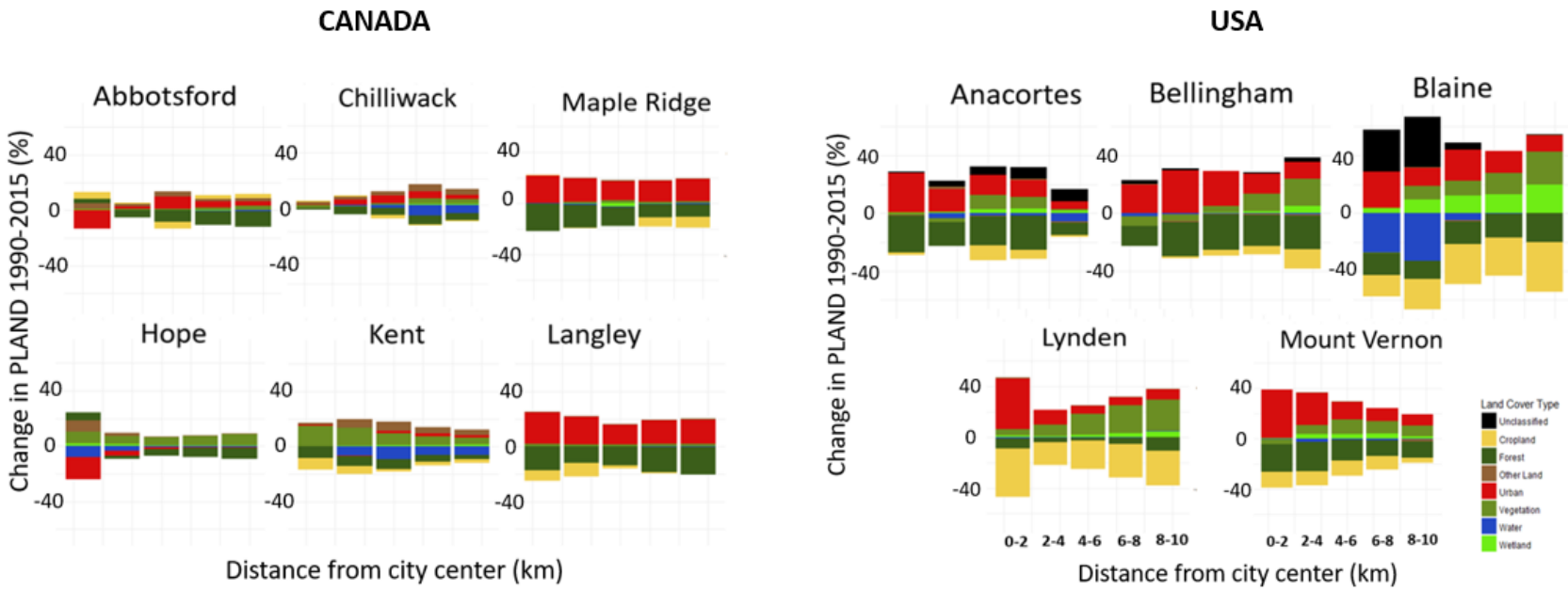

Figure 2.4. Changes in percent or landscape (PLAND) of land cover classes along the rural to urban gradient (0-2 km, 2-4 km, 4-6 km, 6-8 km, 8-10 km radius). Additional information can be found in Appendix Table 2.

In 1990, within a 10 km radius of the city center, forest cover was the most abundant land cover type in eight of eleven cities (Figure 2.3). Forest cover decreased over time (1990 – 2015) in all cities (from 5 to 17% with a mean decrease of 12%). Agriculture was the most abundant land cover type for the remaining cities of Abbotsford, Lynden and Mount Vernon. Urban land cover increased over time in nearly all cities with the exception of Hope, which saw a loss in urban land. Increases in urban cover ranged from approximately 4 to 17% with a mean increase of 13%. Agricultural land declined in nine of the eleven cities, increased in one (Abbotsford), and did not change in one (the city of Hope). The decrease in agriculture ranged from 1% to nearly 27% with a mean decrease of just over 9%. Agricultural land increased by less than 1% in the city of Abbotsford.

2.3.2 Urban land decreased, while agriculture and forest increased with distance from city center

At both timesteps (1990 and 2015), urban land significantly declined with increasing distance from city center (p=0.02) in five cities (Anacortes, Bellingham, Chilliwack, Hope, and Mount Vernon) and marginally declined (p=0.08) along Abbotsford's urban-rural gradient (Figure 2.4, Appendix Table 2). Urban land peaked within a 0-2 km radius from the center of most cities. Overall, the greatest relative increase in urban land occurred within a 0-2 km radius of Lynden and Mount Vernon's city centers (40% and 38%, respectively). The greatest relative loss in urban land occurred within a 0-2 km radius of Hope's city center (16%).

In both 1990 and 2015, agricultural land increased with distance from city center (p=0.02) for the city of Maple Ridge, and marginally increased (p=0.08) in Mount Vernon and Blaine. Agricultural land significantly declined (p=0.02) along Chilliwack's urban-rural gradient (Figure 2.4). In 1990, agricultural land increased along the urban to rural gradient (p=0.02) in

Bellingham and marginally decreased (p=0.08) in Langley, but by 2015 showed no distance trend in either city. In contrast, while agricultural land surrouding Anacortes and Hope showed no significant spatial trend in 1990, by 2015 both cities showed marginally significant trends (p=0.08), increasing and decreasing in Anacortes and Hope, respectively.

In 1990, forest increased with distance from city center in five cities (Abbotsford, Chilliwack, and Hope: p = 0.02; Anacortes and Bellingham: p=0.08). By 2015, forest increased with distance from city center in seven cities (Chilliwack: p=0.02; Abbotsford, Anacortes, Bellingham, Hope, Kent and Langley: p=0.08).

2.3.3 Over time, proportional loss of agricultural lands was greater in US than in Canadian cities

Dominant trends in the Canadian versus USA cities were different over time. Urban land cover increased over time in all US and Canadian cities with the exception of the Canadian city of Hope. Four of the five US cities and one-third of the Canadian cities saw a >10% increase in urban land from 1990-2015 (Figure 2.3). All US cities and half of the Canadian cities showed increases over time in urban land in all distance classes. The only cities losing urban land at any distance were the Canadian cities of Kent, Hope, and Abbotsford.

All US cities saw a minimum 5% decrease in agriculture whereas Blaine and Lynden experienced over 25% reduction. Four of the six Canadian cities saw a decrease in agriculture, but only Maple Ridge saw a decrease over 5%. All US and Canadian cities demonstrated an overall loss in forest and the greatest relative decrease in forest cover occurred in the US cities of Bellingham, Anacortes, and Mount Vernon and the Canadian city of Langley.

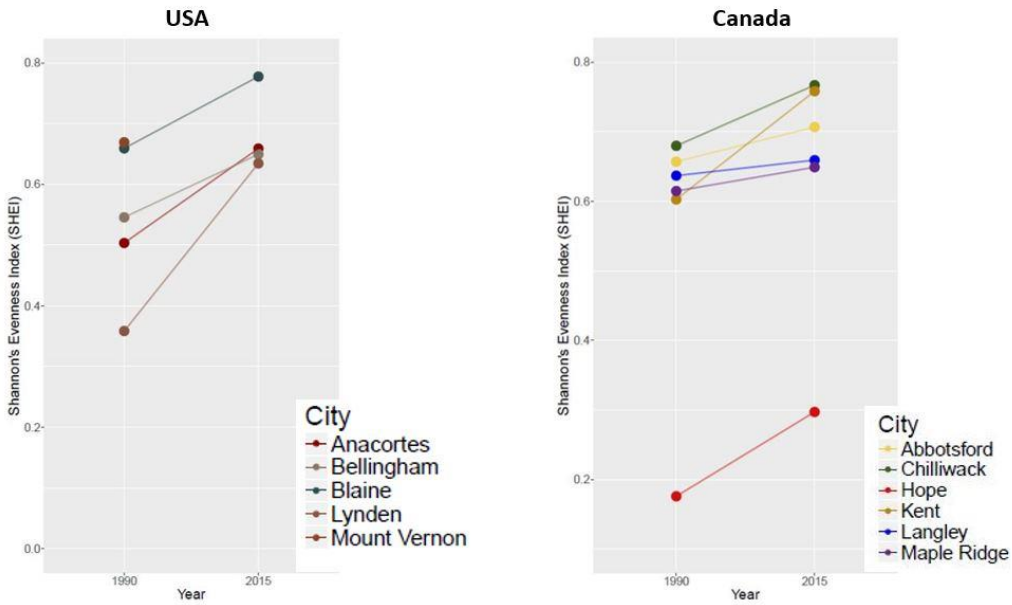

Figure 2.5. 1990 and 2015 SHEI for all cities calculated within 10 km radius of the urban center. SHEI increased in all cities from 1990-2015

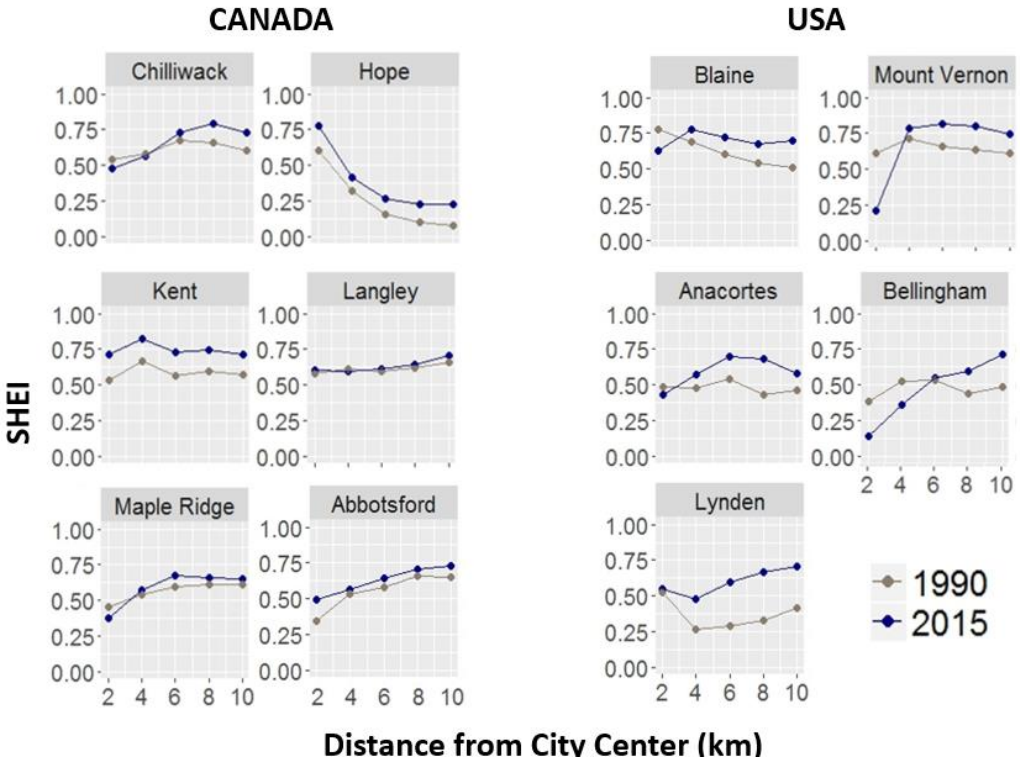

Figure 2.6. Shannon's evenness index (SHEI) for all cities plotted along an urban-rural gradient. Most cities experience peak evenness 2-8 km from the city center.

2.3.4 Landscape Pattern Metrics: SHEI increased in all cities from 1990-2015

From 1990 to 2015, land cover in all cities increased in evenness (Figure 2.5). Changes in SHEI along the urban-rural gradient are shown in Figure 2.6. In 2015, Abbotsford, Blaine, Chilliwack, Kent, and Mount Vernon all exhibited a score of 0.7 or higher indicating that the distribution of patch types were approaching maximum evenness. For the cities of Abbotsford, Bellingham, and Langley, evenness increased with distance from the city center. The opposite was true for Hope, which approached zero with distance from city center. Seven of the eleven cities experienced peak evenness 2-8 km from the city center.

2.4 DISCUSSION

Managing cross-border regions presents an array of political and ecological challenges. Governing bodies and stakeholders may have differing – and sometimes conflicting - agendas. Determining where and how government agencies are working together and identifying areas where cross-border management strategies and policies vary is the first step to creating holistic approaches for managing trans-boundary systems. Understanding the historical and political background of an area helps to inform land managers of which practices are most effective for maintaining a healthy and productive landscape. An important component of this broader challenge of managing cross-border systems is mapping such areas in consistent and comparable way (Pardington and Cardille, 2013). Geodata are routinely collected by a multitude of government and private agencies at differing scales, using various classification schemes, and for varying purposes. This creates many challenges concerning obtaining continuous, uniform, accurate data for a system. Landscape pattern analysis is one way to compare across regions and

jurisdictions, identify how landscape changes are affecting landscape patterns, and in turn, how aquatic resources might be impacted.

2.4.1 Landscape evenness increases as urban land increases and agricultural intensifies in the region

Landscape evenness increased in all cities over the 25-year period examined, driven primarily by gains in urban and losses in agricultural lands. From 1990-2015, urban land increased in all cities while forest and agricultural land decreased by 9% and 12%, respectively, in most cities. Additionally, the Canadian town of Hope was the only city where urban land cover decreased. Agricultural land was lost in all cities except in Abbotsford where it increased slightly. The greatest decreases in agricultural land occurred in the five US cities. While agricultural land decreased over time surrounding most cities, according to the farm census, total production remained the same or increased, indicating the agricultural activities may be intensifying on remaining agricultural lands (USDA 2018, Ministry of Agriculture 2016).

2.4.2 Differences in US-Canadian policy is apparent in landscape patterns

Policy implemented in the US and Canada may drastically differ. Land use policies (zoning, master plans, growth boundaries) help to determine urban form and its impact. For example, in the greater Seattle area, growth management efforts to increase housing densities within growth boundaries has had unintended consequences, encouraging low density housing sprawl in rural and wildland areas just beyond those planned growth boundaries (Robinson et al 2005). The slower loss of agricultural land over time in Canada (as opposed to the severe decline in the US) may be a result of the Canadian Agricultural Land Reserve (ALR). Established in 1973, the ALR is a collection of designated agricultural land in British Columbia. Land parcels falling within the ALR are limited to farm uses according to Agricultural Commissions bylaws.

To remove a parcel from the ALR requires requesting permission from the provincial government. The ALR likely prevents or at least slows the rate of agricultural conversion to urban in the Canadian portion of the study area. Importantly, nearly 70% of the Canadian-side of the Abbotsford-Sumas Aquifer is ALR-zoned land (Figure A.3). As such, understanding how policies impact landscape patterns and identifying where landscape changes are occurring can help to better focus planning efforts to reduce negative consequences of LULC change on aquatic ecosystems.

2.4.3 Identifying changes in landscape pattern helps focus land planning to improve water quality

Landscape patterns serve as useful indicators of overall water quality. The percentage of land in forest and non-forest cover as well as the density of paved roads are among the strongest predictors of overall water quality (Wu and Sun 2016, Hunsaker and Levine 1995, Swank and Bolstad 1994). Additionally, strong links between land cover and water quality changes associated with major storm events suggests that even small changes in land cover have important implications for water quality (Swank and Bolstad 1994, Larson and Grimm 2012, Janke et al 2017). The spatial organization of land cover, measured by contagion and dominance, may also have a bearing on water quality (Hunsaker and Levine 1995, Moreno-Mateos, 2008, Kelting et al 2012). Hence, spatiotemporal changes in landscapes may therefore have important implications for water quality.

Groundwater is an important source of drinking water for the eleven cities in this study. Furthermore these aquifers are shallow, unconfined aquifers highly vulnerable to contamination. As such, understanding patterns of development and land conversion, and linking these changes to implications for water quality, can help planning and regulatory bodies focus efforts on critical

locations that may be negatively impacted by LULC. Within the Greater ASA region, certain areas, mainly the Abbotsford-Sumas aquifer, have experienced the negative consequences of LULC changes. For instance, increases in urban areas have led to an increase in food demand which has resulted in agriculture intensification within the region. This agricultural intensification has increased nitrate contamination of transboundary water sources in the region causing an international dispute over water quality. Mapping changing landscape patterns can help identify where land use is most likely to be intensified, or conversely, where it will likely remain stable or change in only trivial ways. In turn, this information can be used to create indicators of potential contamination and used in conjunction with water quality data to map and model areas vulnerable to contamination. In a world of limited resources, such localized targeting may be as or even more effective than broad regulations intended to protect water quality (Wear et al 1998).

2.5 CONCLUSIONS

Urbanization and agricultural intensification modify landscapes producing numerous unintended consequences. My results showed that in most cities in the Greater ASA area, urbanization is encroaching upon agricultural land use, which in turn is encroaching upon forested lands. Furthermore, patterns of urbanization and agricultural intensification along the urban-to-rural gradient can be well captured by landscape metrics even surrounding smaller cities undergoing major landscape transitions. I showed how landscape change is spatially heterogeneous and often occurs differentially along the urban to rural gradient. Additionally, these results provide important knowledge base for land use managers in the region. While this

chapter provides important background context of LULC in the Greater ASA region, the deeper implications of these changes for water quality will be further explored in subsequent chapters.

Chapter 3: Landscape Indicators of Groundwater Nitrate Concentrations: An Approach for Transboundary Aquifer Monitoring

3.1 INTRODUCTION

Nitrate contamination of groundwater is a global concern (Goodchild 1998, Joosten et al 1998, Birkinshaw and Ewen 2000, Saãdi and Maslouhi 2003, Kyllmar et al 2005, Liu et al 2005, Almasri 2007). Globally, groundwater provides approximately 45% of freshwater used for drinking and cooking, and an additional 24% of water used in irrigated agriculture (Van der Gun 2012). Increasing use of synthetic and organic fertilizers, disposal of waste (particularly from animal-based agriculture), and changes in landscape patterns are key factors responsible for the progressive increase in nitrate concentrations in groundwater over the last 30 years (Townsend and Howarth 2010). In response to such increases, the World Health Organization (WHO) has established a maximum threshold of 10mg/L nitrate-N for drinking water to avoid problems such as hypoxemia in infants (Canadian Council of Ministers of the Environment 2014; United States Environmental Protection Agency 2017). Other potential health effects of excess nitrate in drinking water include reproductive problems and high risk of non-Hodgkin's lymphoma (Weyer et al 2001, Weisenburger 1991, Ward et al 1996). Adverse effects may also be possible below WHO guidelines, however; long-term exposure to nitrate in community water supplies as low as 2 - 4 mg N L^{-1} has shown possible links to bladder and ovarian cancer (Weyer et al 2001).

Trans-boundary aquifers are particularly vulnerable to contamination. While nearly 300 river basins traverse international boundaries (Transboundary Water Assessment Programme 2016), twice as many aquifers span international political boundaries (IGRAC 2015). Cross-border aquifers are the primary source of freshwater on almost every continent, yet the number

of international agreements for transboundary rivers and lakes vastly outnumber those of transboundary aquifers (Eckstein and Eckstein 2003). Long water residence time, large storage capacity, physical inaccessibility for remediation, and lack of regulations make many aquifers challenging to manage, especially in cross-border settings (Foster and Chilton 2003). Furthermore, the spatio-temporal scale of data collection and monitoring conducted by different countries may not be compatible nor shared among jurisdictions. Thus, the complexity of coordinating among international agencies working across multiple jurisdictions has also likely exacerbated the long-term difficulties in addressing groundwater nitrate.

Understanding how land use and land cover (LULC) patterns impact groundwater systems is a critical first step towards mitigating nitrate contamination. One approach is through easily measured landscape indicators. Landscape indicators quantify the amount and arrangement of land cover (such as percent agriculture and percent forest cover) and the physical structure of vegetation on the land surface (Meyer and Turner 1994*)*. They allow for an affordable, broad-brush approach to characterizing the landscape and classifying potential LULC impacts. A long-standing, well-developed body of research examines the correlations between landscape indicators and aquatic ecosystems (Gergel et al 2002, Allan 2004, Johnson and Host 2010). The potential mechanisms correlating land cover and water quality are well understood in a qualitative sense; for example, the amount of agriculture in a basin may be associated with higher stream sediment and/or nutrients concentrations (Blake et al 2012, Arheimer and Liden, 2000, Osborne and Kovacic 1993). However, the strength of quantitative predictions can vary greatly depending upon the indicator used and the region in which it is applied.

A plethora of research has examined landscape indicators of surface waters (Hale et al 2004, Mallin et al 2000). However, despite clear connections between surface and groundwater

systems, few studies have examined landscape indicators within the context of monitoring groundwater and aquifers (Gurdak and Qi 2006, Keeler and Polasky 2014). Development of landscape indicators (linked to land use and incorporating topography and geology) can help identify and potentially explain mechanisms and processes acting above and below the land surface (Sophocleous 2002) and are especially relevant to unconfined aquifers which have no overlying impervious rock layer and are therefore susceptible to contamination. Other methods such as such as mass balance nutrient modelling can be expensive and data intensive; however, landscape indicators are a rapid and relatively affordable way to assess likely groundwater contamination. Thus, understanding landscape indicators that relate to groundwater in unconfined aquifers can improve our understanding of terrestrial groundwater interactions.

The Abbotsford-Sumas aquifer (ASA), located in Southwest Canada and Northwest USA, is an ideal location to study this issue. Long-term monitoring (which began in the early 1970s) detected nitrate concentrations in exceedance of WHO standards (Wassenaar et al 2006, Mitchell et al 2003). Decades after being first identified, elevated nitrate concentrations have remained a persistent trans-boundary dilemma for the USA and Canada (Chesnaux et al 2012, Zebarth et al 1998). The complexities of this problem have challenged the many managers, farmers and policy makers who have initiated a wide variety of nutrient management strategies, with little apparent success in reducing nitrate concentrations in the aquifer. Within this context, I ask two primary questions: Are there temporal trends in nitrate concentrations over time? How well do landscape indicators help explain patterns of groundwater nitrate concentrations? To accomplish this, I first tested for statistical trends in nitrate concentrations over time. I then statistically linked agriculturally-focused landscape indicators to nitrate concentrations measured in ASA monitoring wells along the US-Canada border. Based on previous studies, I expect

elevated nitrate concentrations in areas with large amounts of livestock and berry production (Lockhart et al 2013, Wassenaar 1995).

In addition, three subsidiary objectives were examined to determine: a) the spatial scale - or distance - over which landscape indicators should be measured; b) whether incorporating groundwater flow direction into indicators improves model results; and c) whether indicators of nitrate concentrations in US and Canadian differ. To do so, I calculated landscape indicators within terrestrial zones of influence for differently-sized radii surrounding each well and accounted for directionality of subsurface flow. Finally, I compared analyses of landscape indicators and groundwater nitrate concentrations collected at ASA monitoring wells across the US-Canada border. I hypothesize that incorporating distance and directional flow into landscape indicators will increase predictive power and improve our understanding of the factors contributing to high nitrate concentrations throughout the ASA. I further foresee challenges arising from this comparative cross-border approach as geodata and monitoring techniques differ between countries.

3.2 METHODS

3.2.1 Study Site

The 200 km^2 Abbotsford-Sumas-aquifer (ASA) spans the US-Canada border (Figure 3.1) and is situated within the agriculturally productive Fraser-Whatcom Valley. The aquifer supplies drinking water for 100,000 residents of Canada in the city of Abbotsford and the township of Langley (Chesnaux et al 2007) as well as nearly 10,000 people in the United States (towns of Sumas, Lynden, Ferndale, Everson, Nooksack, and scattered rural areas). The unconfined, highly permeable sand and gravel aquifer is recharged primarily by direct precipitation (Fraser Valley Soil Nutrient Study 2005) and consists of mostly coarse-grained sediments of glaciofluvial drift

origin. Such loose, or unconfined, soil strata provide minimal filtration for contaminants (USGS 1999). Mean groundwater age is approximately 20 years old while models suggest a mean travel time of 6.3 years to reach a position 10 m below the water table when travelling by advection from the top of the water table (Chesnaux et al 2012). Red raspberry is the predominant agricultural crop in the region followed by significant areas of forage grass and pasture (Zebarth et al 2015).

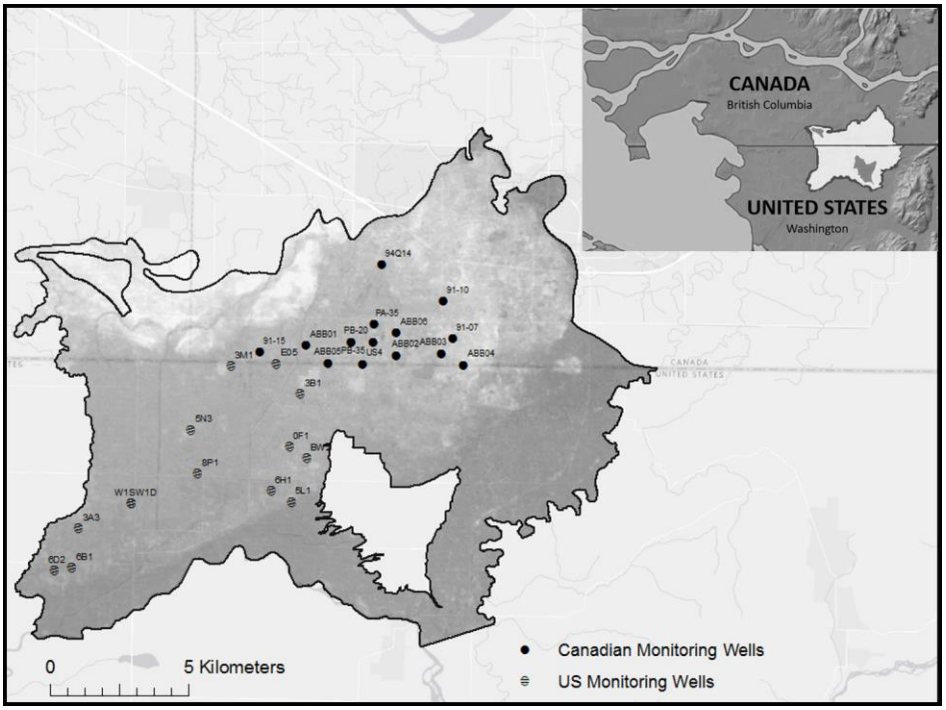

Figure 3.1. The Abbotsford-Sumas study region located in southwestern British Columbia and northern Washington is a 200 km2 unconfined, highly permeable sand and gravel aquifer recharged primarily by direct precipitation. 15 shallow groundwater monitoring wells were located in the Canadian side of the aquifer while 14 were located on the US side of the aquifer. (*Modified from a map originally from: Martin Suchy, Environment and Climate Change Canada*).

3.2.2 Groundwater Nitrate Monitoring

I examined nitrate concentrations from 15 shallow groundwater wells in Canada and 14 in the US portions of the ASA. Available groundwater nitrate measurements [expressed in mg N

L^{-1}] collected between 2005-2013 were obtained from Environment and Climate Change Canada (ECCC) and the State of Washington Department of Ecology (Figure 3.1). Because shallow groundwater is younger (and therefore more likely to reflect more recent landscape practices than deeper groundwater), only shallow wells with mid-screen depths less than 10m below the mean height of the water table were used. Prior to June 2013, ECCC sampled monthly; however to reduce program costs, they began quarterly sampling thereafter (i.e. in March, June, September, and December). For Canadian wells, mean, median, minimum, and maximum nitrate were determined on a quarterly and annual basis. Many US wells had only one sample collected between 2005-2013.

3.2.3 Geospatial Data

To assess contemporary (2012) land use and land cover, a variety of geospatial datasets were consolidated from US and Canadian sources (Table 3.1). The BC Ministry of Agriculture's Canadian Agricultural Land Use Inventory (ALUI) (2012) contains detailed, hierarchically organized agricultural information depicted as polygons. The ALUI does not cover the entire Canadian-ASA region, however. Thus, to fill gaps in urban areas, the Metro Vancouver Multi-Spectral (MVMS) dataset (2010) was used. Together these datasets provided the best continuous coverage for the Canadian portion of the ASA region. In the US portion, both agricultural and urban land cover were obtained via the United States Department of Agriculture (USDA) National Agricultural Statistics Service (NASS) Cropland Data Layer (CDL) (2012). To create a seamless cross-border mosaic, these datasets were harmonized to commensurate resolutions, formats, and classification schemes representing LULC categories consistent across the region.

Table 3.1. Comparison of characteristics of geodatasets from the US and Canada used in this chapter. Approach for improving concordance among these datasets is explained further in the text.

Geodataset	Source	Year	Extent	Data Format	Spatial Resolution	Number of Classes	Description of Classes
Agricultural Land Use Inventory (ALUI)	Canadian Ministry of Agriculture	2012	CAD	polygon	500 m^2 or width=10m	>200	Detailed information describing agriculture land uses with many hierarchical subclasses (e.g., Anthropogenic - Terrestrial - Vegetated - Cultivated - Barley)
Metro Vancouver Multi-Spectral (MVMS)	Metro Vancouver Regional District	2010	CAD	polygon	10 m	16	Urban classes (e.g., roads, built environments, urban mixed)
National Agricultural Statistics Service (NASS) Cropland Data Layer (CDL)	US Dept. of Agriculture (USDA)	2010	USA	raster	30 m	28	Generalized agricultural classes with few subclasses (e.g., non-natural woody - orchards/vineyards/others)

3.2.4 Landscape Indicators

Understanding the relationship between the land use near a well and contaminants in groundwater provides insights into possible sources of contamination. To determine linkages between LULC and nitrate concentrations, contemporary (2012) land use patterns (e.g. proportion and abundance of land cover types) were used in the creation of a variety of landscape indicators (Table 3.2) measured surrounding each groundwater stations. Building on previous studies using multiple-sized radii surrounding wells, I calculated these landscape indicators within adjacent "zones of influence" surrounding monitoring wells (Keeler and Polasky 2014, Burow et al 2010, Nolan et al 2002, Frans et al 2012) described further below.

In addition to the landscape indicators in Table 3.2, I also examined the recent history of raspberry field renovations (capturing both replanted raspberries as well as new raspberry fields) surrounding each well. Due to build-up of root pathogens and viruses in soil, raspberry fields are typically renovated (uprooted and replanted anew) every 5-8 years. Renovations include removal of root balls and old canes followed by tillage. A critical component of renovations, aside from planting of new raspberry plants, is the addition of soil amendments (typically poultry manure) to increase soil nutrients. Renovations occur in the fall and the following spring manure is applied to bare fields and new raspberries are planted (Forge 2012). It is during this application of manure in the spring when fields are bare that it is hypothesized intense leaching occurs (Staver and Brinsfield 1998). In order to capture the dynamic nature of this regionally-abundant crop, Google Earth imagery was used to photo-interpret past renovations of raspberry fields (i.e. field turnover), since 2004 within 100 m semi-circular zones of influence in the Canadian portion.

Table 3.2. Landscape indicators and additional co-variates were calculated within concentric zones of influence (100, 500 m) surrounding each groundwater monitoring well.

Landscape Indicator	Units	Description	Hypothesized Relationship to Groundwater Nitrate
% Berries	Proportion of Zone	Area used for berry production. Further distinction was made among berry types: raspberries only, blueberries only, as well as mixed berries (areas with a mix of berry types – raspberries, blueberries, strawberries, cranberries, and blackberries).	Southern BC is known as the "raspberry capital of the world." Other types of berries are grown in the region. Berry production has been associated with high fertilizer use in the region.
Berry field renovations	Total Area of Renovations (m^2)	Newly-planted and replanted raspberry fields (m^2) since 2004	Raspberries are fertilized heavily within the first 3 years of planting with declines in fertilizer application in subsequent years.
% Forage / Pasture	Proportion of Zone	Active (and inactive) forage and pasture lands for livestock.	Manure is a source of nitrate.
% Other Agriculture	Proportion of Zone	Areas of land not classified as other types of agriculture listed above, including vegetable crops in the area such as maize.	Throughout the ASA, the main source of nitrate to the aquifer is likely attributed to agriculture (Zebarth et al.1998, Chesnaux et al 2012).
% Forest	Proportion of Zone	Trees (deciduous, evergreen, and mixed). Land with >10% vegetation cover	Natural vegetation has been shown to buffer impacts of nitrate contamination.
% Urban	Proportion of Zone	Areas with a mixture of constructed materials and vegetation (e.g., roads, lawns, barns, houses).	Impervious surfaces (such as roads and pavement) can increase runoff and fertilized lawns are potential sources of nitrate
% Bare Land	Proportion of Zone	Bedrock, gravel pits, and other accumulations of earthen material. Generally, vegetation accounts for <15% of total cover.	Large gravel mining operations in the ASA may contribute to surface and groundwater pollution. Impervious surfaces are not included in this category as they are included in urban.
% Wetland	Proportion of Zone	Area with forest or shrub land vegetation where the soil or substrate is periodically saturated or covered with water.	Denitrification in wetlands potentially reduces nitrate loading to surface and subsurface waters.
% Water	Proportion of Zone	Open water, generally with < 25% vegetation or soil	Streams and ponds impact surface-groundwater recharge zones, potentially affecting nitrate concentrations.
Additional Co-variates			
Water Table Height	m	Mean height of the water table 1996-2013	As water tables rise, increase leaching of nitrate to groundwater may be expected.
Depth of Mid-Screen	Depth below water table (m)	The mid-screen refers to the location in the well where water enters. The depth of this mid-screen determines depth (below the average height of the water) for water sampling. I used mean depth of screen casing from 1996-2013.	Depth is related to the movement of groundwater and is associated with the age of groundwater. Hence, it can be expected that the deeper the mid-screen, the older the water being collected.

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The above landscape indicators were measured in zones of different shape (circular vs. semi-circular) and increasing radii (100 and 500 m) to identify the scale and directionality of the strongest correlation between land use and nitrate concentrations. Because groundwater moves both laterally and vertically in an aquifer (USGS 2005), radii of different sizes were evaluated to capture potential lateral water movement. Zones of 1 km radii were also evaluated but due to lack of statistical significance were not included here. I further incorporated a unique approach using upstream semi-circular zones of influence corresponding to the predominately-southwesterly direction of groundwater flow (Figure 3.2). While the zones of influence do not address residence time or aquifer volume (both of which may be important to groundwater nitrate concentrations), they do help further explore directionality as well as the extent of landscape influence. Additional co-variates (i.e. water table height and average depth of screen for each well) were calculated and incorporated into models (further described in Table 3.2).

Figure 3.2. Landscape indicators within "upstream" terrestrial zone of influence. Semi-circular shaped zones (from 100 m and 500 m surrounding each well) were created to incorporate the southwesterly groundwater flow direction.

3.2.5 Statistical Analysis

First, to examine long-term trends in nitrate concentrations, I used Mann-Kendall (MK) tests to detect either monotonic upward or downward trends over time (from 2005-2013). A monotonic trend upward (or downward) means that a variable consistently increases (or decreases) over time, yet the trend may or may not be linear. A positive MK score indicates an increase with time whereas a negative indicates the opposite. Sen's slope is used to estimate the rate of change of a trend (Hersel and Hirsch 1992) and permits a comparison of the strength of correlation between two data series (Hirsch and Slack 1984). Though Mann-Kendall tests can be

computed for a time series with missing values, performance will be adversely affected. Therefore, missing values were interpolated using the mean value of the two years prior and two years after any missing quarterly samples. Measurements collected in January 2007 replaced missing values in December 2006. Wells with more than 3 missing values throughout the time series were excluded. Long-term trend analysis was only conducted for Canadian wells as comparable long-term nitrate measurements were not available for US wells. Trend significance was mapped for each well.

My second approach examined connections between nitrate concentrations and landscape indicators. For wells located in Canada, measures of central tendency (mean, median) as well as minimum and maximum nitrate were determined annually as well as quarterly (March, June, September, and December). Sen's Slope (from the Mann-Kendall tests above) was also used as a dependent variable for wells in Canada. Annual median nitrate and Sen's Slope showed the strongest correlation with landscape indicators (using exploratory Pearson correlation coefficients) and were chosen as the dependent variables for further analysis. For US wells (only sampled once within the study period), a single nitrate sample value was used as the response variable.

Backward stepwise regression was used to determine which landscape variables (Table 3.2) measured over four scales (circular and semi-circular, each with 100 m and 500 m radii) were significant in models predicting nitrate concentrations. My goal was to seek the best models consisting of no more than 2-4 independent variables to avoid model over-fitting. To accomplish this, the least significant variables in each model were sequentially omitted in the interest of parsimony (as judged by their significance levels, partial R^2 values, and AIC scores). Akaike information criterion (AIC) aids model selection by evaluating the relative quality (or goodness-

of-fit) of different statistical models and helps identify and penalize models which are over-fit with too many additional variables (Burnham and Anderson 2003). Models with the lowest AIC score, given a similar number of independent variables, indicate the highest quality model. Additional variables were warranted in a model only if they lowered the AIC by at least two points. Only significant models (p-value <0.1) were reported.

Shapiro-Wilk tests were used to assess normality of independent and dependent variables and landscape indicators were transformed (as needed) using the arcsine square root transformation. Collinearity among independent variables was assessed using Pearson correlation coefficients and highly correlated landscape indicators (greater than r = 0.80) were not included in the same model.

3.3 RESULTS

3.3.1 60% of wells showed decreasing trends in nitrate

Wells in Canada had been sampled >70 instances between 2005 and 2013 and nitrate concentrations ranged from 1.3 - 61.6 mg N L^{-1}. Mann-Kendall tests were significant ($p < 0.1$) for eleven of the fifteen Canadian wells. Nitrate concentrations in nine wells decreased from 2005-2013 (Figure 3.3). Two wells demonstrated a significant increasing trend over time (Figure 3.4). The remaining four wells demonstrated no significant trends. Figure 3.5 shows locations of

wells noting increasing and decreasing trends. Recall that trends could not be evaluated for US wells because long-term nitrate measurements were not available.

Figure 3.3. Of the 16 Canadian wells examined, 9 exhibited significant declining trends in nitrate concentrations from 2005-2013 (Mann-Kendall tests with p = ≤ 0.1 significance threshold

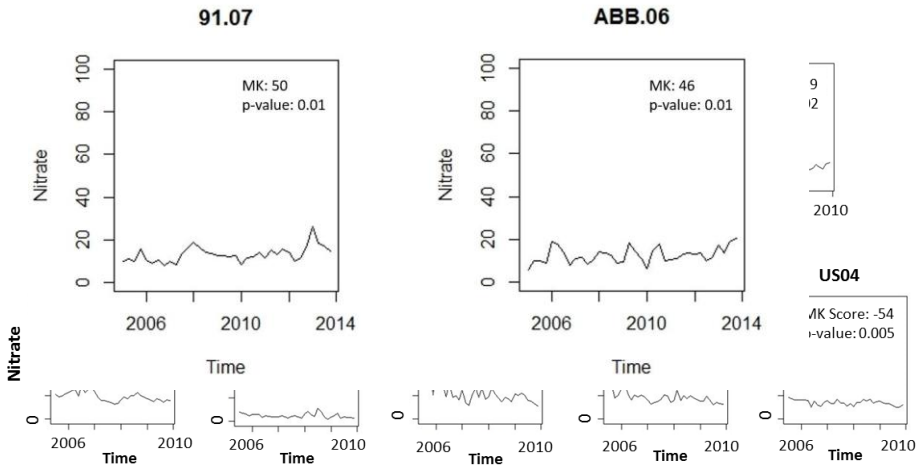

Figure 3.4. Two Canadian groundwater monitoring wells exhibited increasing trends in nitrate concentrations over time (2005-2013) using Mann-Kendall tests with (p = ≤ 0.1).

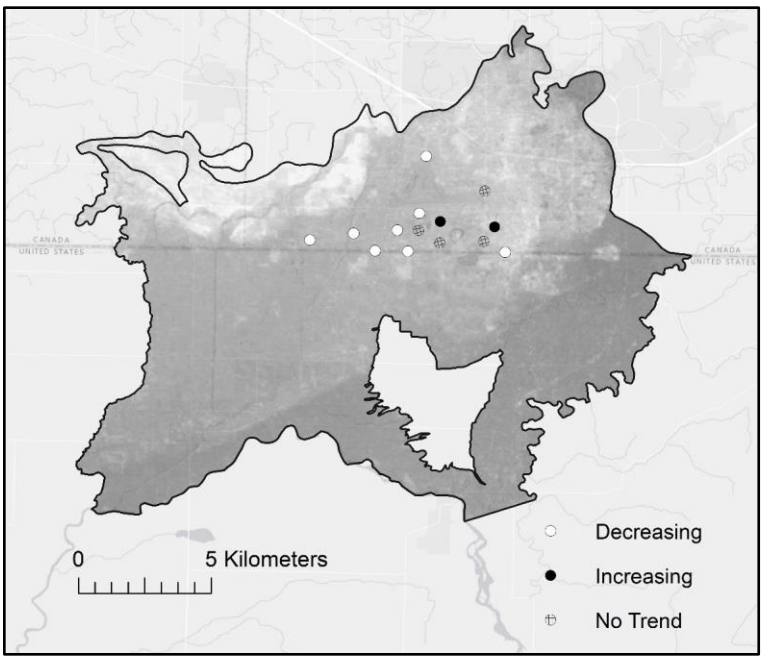

Figure 3.5, Direction of significance of trends in nitrate concentrations (2005-2013) in Canadian wells according to Mann-Kendall tests (p = ≤ 0.1)

Next, I examined connections between water chemistry and landscape indicators. The strength of Pearson correlation coefficients indicated that annual median nitrate concentrations and Sen's Slope (derived from M-K tests) were most correlated to landscape indicators and thus most suitable for further analysis. In contrast, the lone nitrate sample available for each US well (ranging from 0.013 - 34.1 mg N L^{-1}) was used as the response variable for statistical analyses of US wells.

3.3.2 Proportion of raspberries as well as forage and pasture land are important predictors of groundwater nitrate concentrations

A total of 12 models were created (4 scales of influence x 3 response variables): Sen's Slope (Table 3.3A), annual median (Table 3.3B) and US nitrate (Table 3.4), achieving full model R^2 as high as 0.72. The best model for each response variable is shown in bold (Table 3.3 and

Table 3.4). Regardless of zone radii size, direction, or jurisdiction, the proportion of raspberries and forage and pasture most strongly and consistently explained groundwater nitrate concentrations. Of the variables positively related to nitrate concentrations in Canada, proportion of forage and pasture land routinely explained the most variance (partial R^2 = 0.05-0.29) followed by area of renovations (partial R^2 = 0.11 – 0.25) and other agriculture (partial R^2 = 0.11 – 0.15).

Table 3.3. Models assessing nitrate concentrations on the Canadian-side of aquifer using Sen's Slope (A) and annual median nitrate (B) a response variable (100 m and 500 m, circular and semi-circular shaped). Models in bold represent best model for that response variable based on Adj. R^2 and AIC values. Significance levels: '*' p = ≤ 0.1, '' p = ≤0.05, '***' p= ≤ 0.01.**

(A) Sen's Slope (n = 15)

Zone of Influence	**Model R^2**	**Adj. R^2**	**p-value**	**AIC**	**Independent Variables** (and direction of relationship)	**Relative Importance** (Partial R^2)
100 m Circular	0.43	0.28	0.08	27.92	+ Raspberries**	0.16
					+ Other Agriculture**	0.15
					+ Depth of Screen*	0.11
500 m Circular	0.42	0.26	0.09	28.28	+ Raspberries*	0.21
					+ Managed Vegetation**	0.15
					+ Forage and Pasture	0.05
100 m Semi-Circular	0.59	0.43	0.04	24.88	+ Forage and Pasture**	0.29
					- Blueberries**	0.10
					- Urban**	0.10
					- Mixed Berries	0.09
500 m Semi-Circular	**0.63**	**0.49**	**0.02**	**23.20**	**+ Forage and Pasture**	**0.23**
					- Blueberries	**0.14**
					- Bare Land	**0.14**
					+ Other Agriculture	**0.11**

(B) Annual Median Nitrate (n = 15)

Zone of Influence	Model R^2	Adj. R^2	p-value	AIC	**Independent Variables** (and direction of relationship)	**Relative Importance** (Partial R^2)
100 m Circular	0.51	0.31	0.09	91.60	- Water Table Height**	0.18
					+ Other Agriculture *	0.11
					+ Depth of Screen**	0.11
					+ Urban*	0.10
500 m Circular	0.52	0.33	0.08	91.27	+ Mixed Berries***	0.34
					+ Blueberries*	0.07
					+ Forage and Pasture*	0.07
					+ Depth of Screen	0.02
100 m Semi-Circular	**0.72**	**0.61**	**0.007**	**82.97**	**+ Area of Renovations ** **	**0.25**
					+ Forage and Pasture* **	**0.21**
					**- Bare Land* **	**0.13**
					**- Raspberries* **	**0.12**
500 m Semi-Circular	0.59	0.42	0.04	89.00	- Forest	0.32
					+ Area of Renovations	0.11
					+ Depth of Screen	0.08
					- Raspberries	0.06

Table 3.4. Comparison of models for assessing nitrate concentrations on the USA-side of aquifer using indicators measured within 100 m and 500 m, circular and Semi-circular shaped zones of influence. Models in bold represent best model for that response variable based on Adj. R^2 and AIC values. Significance levels: '*' p = ≤ 0.1, '' p = ≤0.05, '***' p= ≤ 0.01. (n = 14)**

Zone of Influence	**Model R^2**	**Adj. R^2**	**p-value**	**AIC Score**	**Independent Variables** (and direction of relationship)	**Relative Importance** (Partial R^2)
100 m Circular	0.21	0.14	0.09	101.6	+ Raspberries*	0.21
500 m Circular	**0.45**	**0.35**	**0.03**	**98.44**	**+ Forest***	**0.37**
					+Urban	**0.07**
100 m Semi-Circular	N/A	N/A	N/A	N/A	N/A	N/A
500 m Semi-Circular	N/A	N/A	N/A	N/A	N/A	N/A

In explaining median annual nitrate, area of renovations and proportion of forage and pasture were positively associated with nitrate in half of the models, with partial R^2 values ranging from 0.11-0.25 and 0.07-0.21, respectively. Proportion of mixed berries was positively associated with nitrate values in one model with a partial R^2 value of 0.34. In contrast, proportion of raspberries was weakly negatively associated with nitrate in half of the models, with partial R^2 values ranging from 0.06-0.12. Proportion of forest and water table height were negatively associated with nitrate in one of the models each, with partial R^2 values of 0.32 and 0.18, respectively.

In explaining the Sen's slope, which indicates rate of change in a trend over time (Helsel and Hirsch, 1992), proportion of raspberries, forage and pasture, and other agriculture were positively associated with nitrate trends at least half of the models, with partial R^2 values ranging from 0.16-0.21, 0.05-0.29, and 0.05-0.29, respectively (Table 2.3A). In contrast, the proportion of blueberries were negatively correlated with Sen's slope in half of the models with partial R^2 ranging from 0.10-0.14

3.3.3 Larger 500 m zone radii improved most models

Using a larger zone of influence to measure landscape indicators improved both US and Canada models. Larger (500 m) zones of influence improved R^2 values for US models by more than double (R^2=0.45 for 500 m circular vs. R^2=0.21 for 100 m circular) (Table 3.4). In Canada, models of Sen's Slope were improved somewhat by using larger zones of influence (R^2=0.63 for 500 m semi-circular zones vs. R^2=0.59 for 100 m semi-circular zones) (Table 3.3A). In contrast, models of annual median nitrate were improved by using smaller zones of influence (R^2=0.59 for 500 m semi-circular zones vs. R^2=0.72 for 100 m semi-circular zones) (Table 3.3B). For all other models, changing the size of the zone of influence did not significantly change R^2 values.

3.3.4 Incorporating flow direction was beneficial regardless of scale

Incorporating flow direction by incorporating semi-circular zones of influence returned significant results for all Canada models, but not for the US. For models of Sen's Slope, incorporating direction greatly improved models using landscape indicators at 100 m (R^2=0.59 for semi vs. R^2=0.43 for circular zones) as well as at 500 m distances (R^2=0.63 for semi vs. R^2=0.42 for circular zones) (Table 3.3A). For models of annual median nitrate, incorporating direction greatly improved models at 100 m (R^2=0.72 for semi vs. R^2=0.51 for circular zones) and slightly improved models at 500 m (R^2=0.59 for semi vs. R^2=0.52 for circular zones) (Table 3.3B).

3.4 DISCUSSION

3.4.1 Landscape indicators are useful predictors of groundwater nitrate

My results are consistent with previous studies reporting significant positive relationships between nitrate concentrations in shallow aquifers and agricultural land surrounding wells (Keeler and Polasky 2014, Tesoriero and Voss 1997, Eckhardt and Stackleberg 1995). My results show nitrate concentrations were strongly correlated (e.g. partial R^2 = 0.11-0.29) with the proportion of forage and pasture land, and proportion of raspberries surrounding wells. The strength of these correlations supports the premise that land use affects groundwater quality in aquifers overlain by highly permeable materials. Previous studies using an allocation model for groundwater nutrient loads based on land cover classes, found similar results determining that average nitrate concentrations were highest beneath cropped fields and residential areas (Schilling et al 2016). Other studies have found higher nitrogen concentrations in groundwater in watersheds dominated by agricultural as compared to forestry-dominated catchments (Lawniczak et al 2016).

The strongest models of nitrate consistently suggest that area of renovations, forage and pasture land as well as proportion of berries explains as much as 72% of nitrate concentrations. Yet, berries were both positively and negatively related to nitrate concentrations, depending on type (mixed, blueberries, and raspberries), response variable (annual median nitrate or Sen's Slope) and jurisdiction (US or Canada). In Canadian wells, for models using Sen's slope of nitrate trends as a response variable, proportion of raspberries was strongly positively correlated with nitrate; while proportion of blueberries was weakly negative. For models of annual median nitrate, mixed berries were strongly (and blueberries weakly) positively correlated with nitrate while raspberries demonstrated a negative relationship.

In contrast, on the US-side of the aquifer where groundwater data were largely lacking, raspberries were positively correlated with nitrate, albeit weakly. Raspberries were the only berry type on the US-side to demonstrate a significant relationship of any kind with nitrate. This shift from positive to negative correlation with nitrate may be a result of varying agricultural management practices between the two countries over time or the fact that blueberries/mixed berries represent less than <14% of all berries grown on the US side of the aquifer as compared to the Canadian side where they represent nearly half of all berries grown. Reduced sampling effort (frequency of observations collected) on the US side may have also played a role and certainly would not represent temporal trends.

3.4.2 Spatial scale of measurements

An emphasis on LULC in circular areas surrounding water table wells is a simple and effective method for correlating land use and water quality (Barringer et al 1990). I examined two spatial extents by measuring landscape indicators within 100 and 500 m distances surrounding wells. Increasing the zone of influence improved model fit for two of the three

response variables. One might reasonably expect 100 m zones of influence to improve model fit because landscape activities immediately surrounding groundwater wells might influence recorded nitrate concentrations in wells more than activities further away, however my results indicated the converse. One explanation is that the well screen depth of the wells sampled is sufficiently below the water table to capture water originally infiltrating further up-gradient than 100 m (Zebarth et al 2015). The 100 m scale of measurement slightly improved model fit for annual median nitrate, however.

The radius size used in the literature has varied greatly, but is an important consideration. For larger radii, land within its perimeter contributes proportionally less water to a well, potentially weakening correlations between groundwater quality and land use. However, a smaller radius, approaching the size of the minimum mapping unit, might unduly influence derived landscape indicators via exclusion of important up-gradient features, susceptibility to localized positional errors in mapped features, or misidentification of fine-scale features within the zone. Thus, selection of an appropriate zone of influence is important to maximizing the correctness of an association between land use and groundwater quality (McLay et al 2001). Similar to my findings, other studies have found significant positive relationships between shallow aquifer nitrate concentrations and land use in estimated recharge zones within a 500 m radius of wells (Keeler and Polasky 2014, Nolan et al 2002, Kolpin 1997, McLay et al 2001). One study reported a radius between 100-250 m as most appropriate (Barringer et al 1990) while others suggest a much larger >3 km radius around wells to explain elevated nitrate concentrations (Tesoriero and Voss 1997).

3.4.3 Landscape indicators help fill gaps in sparse information on nutrient management practices

Nitrogen loading from synthetic fertilizer and manure is an important component of assessing potential sources of nitrate contamination (Keeler and Polasky 2014, Nolan and Hitt 2006, Nolan et al 2002). My incorporation of berry field renovations (associated with greater fertilizer application) was a unique component of my landscape indicator approach and helps improve mechanistic understanding of potential N sources. No prior studies have explicitly examined the number or areal extent of field renovations in this way, despite being a suspected source of nitrate. That field renovations showed a strongly significant positive correlation with annual median nitrate concentrations is consistent with renovated fields being heavily amended with poultry manure before being replanted with raspberries.

In the Canadian portion of my study region, the Environmental Farm Plan (EFP) program initiated by Agriculture and Agri-Food Canada (AAFC) in partnership with BC Ministry of Agriculture supports the development of nutrient management plans that outline nutrient management practices for individual farmers. However, these nutrient management plans are voluntary and proprietary with indications of minimal farmer “buy-in” throughout the aquifer region. As a result, information regarding N loading is largely unavailable for this region. Instead, crop types within a certain distance of wells was my proxy for N loading. Although knowledge of crop type at a location helps indicate where inorganic and organic fertilizer is likely to be applied, it does not indicate the rate of actual application. The rate and timing of N fertilizer application vary based on regional and local factors including crop type, tillage practice, crop rotation, and irrigation practices and from farmer to farmer. The relationship between these regional and local factors along with nitrogen use efficiency results in residual soil

nitrogen being susceptible to leaching into the aquifer after heavy periods of precipitation, despite the lack of direct measurements.

3.4.4 Additional factors

Factors not included in this study may help further explain nitrate in the aquifer. Additional sources of nitrate to groundwater may include septic tanks, lawn fertilizers, and domestic animals in residential areas (Nolan et al 1998). Nitrate concentrations in groundwater on Nantucket Island, Massachusetts, increased with number of septic tanks and percentage of high-density residential and agricultural land and decreased with percentage of forest and undeveloped land (Gardner and Vogel 2005). However, in my study region, studies indicate minimal influence of septic tanks (Robertson et al 2016). Geological factors such as soil drainage type help explain nitrate concentrations (Nolan et al 2002). However, within the relatively small study area (~200 km^2) examined here, there is little variation in soil type and geology among well locations (Soils Landscapes of Canada 2012). Future research should take into account livestock production (both poultry production present in Canada and dairy production present in the US) across the aquifer. Additionally, incorporating past land use into the models would help to capture lag-effects and account for legacy nitrate present across the aquifer. In the next chapter, I build off these results and address the persistence of landscape legacies in impacting measured groundwater nitrate concentrations.

3.4.5 Lack of consistent data for evaluating transboundary systems is a global problem

One major challenge of cross-border monitoring can be a lack of consistent data among governing jurisdictions. For example, sampling frequency of groundwater nitrate varies greatly between the US and Canada. ECCC in British Columbia has a network of monitoring stations spread across the Canadian side of the ASA in operation since the 1970s. The US has monitoring

stations in Whatcom County, WA, managed by a myriad of individual landowners, private and governmental agencies. This inconsistent frequency of sampling makes it challenging to compare long-term groundwater nitrate concentrations between the US and Canada. Further exacerbating cross-border monitoring is the lack of consistency in monitoring within each country. As mentioned, ECCC began domestic well monitoring in the 1970s, and started the dedicated network program in the late 1980s. When the program started, samples were collected monthly. To cut down on program costs, quarterly sampling was implemented in June 2013. This lack of temporal uniformity introduces challenges when statistically analyzing long-term trends, as does singular sampling, as available for the USA portion.

A lack of consistent transboundary information of relevance to managing groundwater is an issue of global proportions. Approximately 40% of the world's *population lives in river*/lake *basins* comprising two or more countries (United Nations 2014). The political dimension of water becomes increasing important when shared across national boundaries and can be a potential source of conflict (Mylopoulos and Kolokytha 2008). In contrast, shared water resources can also provide opportunities for discourse leading to cooperation leading to joint management and monitoring. In addition, a lack of integrated approaches and legal agreements as well as administrative shortcomings, make transboundary cooperation and management difficult (Rahaman and Varis 2005, Katerere et al 2001).

Countries monitoring the same aquatic system may have different goals and thus different intended applications for their data collection. As such, comparison and integration of data is often hampered by a myriad of issues including the spatio-temporal resolution of data collection, lag time issues in transport/detection from point of entry into the system, as well as the types of variables assessed. In order for transboundary water systems to be adequately managed, joint

approaches and techniques for monitoring should be further developed. Approaches such as landscape indicators, which can integrate existing data from disparate jurisdictions, can potentially play an important role in this integration.

3.5 CONCLUSIONS

Nitrate contamination of groundwater systems, many of which cross international boundaries, is a global concern exacerbated by growing food demands (Goodchild 1998, Joosten et al 1998, Birkinshaw and Ewen 2000). Food demand is projected to increase 59–98% between 2005 and 2050 posing huge challenges for sustainable food production and other ecosystem services produced within linked terrestrial-aquatic systems (Valin et al 2014, Tilman et al 2002). Increasing use of synthetic and organic fertilizers, disposal of waste (particularly from animal farming), and changes in landscape patterns are key factors responsible for the progressive increase in nitrate concentrations in groundwater over the last 30 years (Townsend and Howarth 2010). N fertilizer and manure loading is an important component of assessing potential nitrate contamination (Keeler and Polasky 2014, Nolan and Hitt 2006, Nolan et al 2002).

Globally, between 30-50% of all land is used for pasture or agriculture, making agriculturalists the chief managers of usable lands (Ramankutty et al 2008, Tilman et al 2001). Within the same aquifer, I found nitrate concentrations were increasing and decreasing from 2005-2013 at different wells. Such inconsistent trends in nitrate concentrations across the aquifer could indicate implementation of different nutrient management practices by farmers and/or temporal variability in crop renovation cycles. Educational programs to help guide farmers in the benefits/consequences of nutrient management strategies may help to decrease the overall concentration of nitrate in the aquifer. As such, sharing evidence with farmers as well as policy makers on the crop types and land use practices most statistically linked to nitrate concentrations

is important an important part of finding solutions. Determining best management practices (BMPs), educating land use managers and farmers of those BMPs, and then aiding farmers/providing incentives to implement these BMPs is the first step towards creating a farming culture that can both meet consumer demand while considering aquatic ecosystem services.

Landscape indicators can act as a proxy for N loading and allow for an affordable, broad-brush approach to characterizing the landscape and classifying potential LULC impacts, thus helping to break down the complexity of coordinating among international agencies and helping to address some of the difficulties associated with managing and monitoring groundwater nitrate. My work, creating proxies for nitrate loading to groundwater, provides an important new approach which is transportable to other regions facing similar challenges.

Chapter 4: Historical Land Cover Impacts Contemporary Groundwater Quality

4.1 INTRODUCTION

Prior land use influences ecosystems for decades and even centuries (Foster and Chilton 2003). Particularly problematic are historical land-use practices with lasting impacts on contemporary water quality (Harding et. al. 1998). For example, hazardous waste disposal from industrial activities, acid mine drainage from mining, and chemical leaks from decades old underground storage tanks can have long-term repercussions on aquatic ecosystems (Bhaduri et. al. 2000). Agricultural operations are among the primary sources of nonpoint source (NPS) pollution to aquatic systems. Excess nutrients from fertilizer and manure, particularly excess nitrogen (N) and phosphorus (P), have had enormous consequences to freshwater systems globally. The Chesapeake Bay, the Mississippi, inland and coastal waterways of Florida, and the Great Lakes are among many systems with persistent histories of water quality degradation and eutrophication (Dale et. al. 2010, USEPA 2011, Sharpley et. al. 2012). Despite our knowledge of landscape legacy effects on surface waters (Sharpley et. al. 2014, Foster and Chilton 2003), few studies examine landscape legacies impacting groundwater.

Decades of intensive agriculture has led to groundwater nitrate contamination worldwide. Due to the high mobility of nitrate (NO3-), groundwater is particularly susceptible to contamination from leaching, especially in shallow unconfined aquifers underlying agricultural lands. Potential health effects of nitrate contamination in drinking water include blue baby syndrome and increased cancer risk (Weyer et. al. 2001, Weisenburger 1991, Ward et. al. 1996). Additionally, environmental N-loading can contribute to loss of habitat in aquatic systems

(Johnson et. al. 2010, Schindler 2006, Smith et. al. 2006, Howarth et. al. 2000). Since the 1950s, increased fertilizer use, in combination with nitrogen fixation by crops, mineralization of animal manure, and various other sources has resulted in increased release of N into the environment (Puckett et. al. 2011, Galloway et. al. 2003, Barbash and Resek 1996). In the last 60 years, the use of industrially fixed N in the form of fertilizer increased 20-fold in the United States (Puckett et. al. 1995). Depending upon soil type, irrigation, climate, and crop type, as much as 45% of N applied to fields is lost to groundwater through runoff or NO3- leaching (Kros et. al. 2011).

The movement of contaminants such as nitrogen in groundwater adds a layer of complexity to understanding the interaction between land use practices and water quality. Though nitrate is highly mobile and quickly leaches to groundwater, the residence time of water in aquifers is typically several orders of magnitude higher than in lakes and wetlands (Philips et al 2016). Nitrogen in groundwater moves laterally as well vertically and as such can take years (or decades) to move deep into an aquifer. Shallow groundwater (closer to the surface) is younger and more likely reflects recent landscape practices than deeper water. In the US and Canada, shallow wells are typically used for agricultural irrigation, whereas deeper wells are used for private water supplies, and the deepest wells are used for public water supplies. Eventually, as nitrate moves deeper into an aquifer, impacts to drinking water supplies can occur. The variable travel-time of nitrate through groundwater systems creates a lag-time between when nitrogen is first applied onto the land surface and when it is captured by groundwater monitoring wells at various depths. As such, understanding impacts of land use and land cover (LULC) change on groundwater systems is a critical first step in managing nitrate.

Landscape indicators are one approach to quantifying and understanding the impacts of LULC change on aquatic systems. Landscape indicators quantify the amount and arrangement of

land cover (such as percent agriculture and percent forest cover) on the land surface (Meyer and Turner 1994) and are a cost-effective approach to characterizing regional change. A plethora of research has examined landscape indicators of surface waters (Hale et. al. 2004, Mallin et. al. 2000) including a well-developed body of research examining landscape indicators of stream/riverine condition (Karr and Chu 2000, Gergel et. al. 2002, Allan 2004, Johnson and Host 2010). There is a clear understanding of the potential mechanisms correlating land cover and water quality in a qualitative sense; for example, the amount of agriculture in a basin may be associated with higher stream sediment and/or nutrients concentrations (Blake et. al. 2012, Arheimer and Liden 2000, Osborne and Kovacic 1993). However, development of richer detail in landscape indicators that potentially identify and explain mechanisms of water quality pollution, both above and belowground, is needed (Sophocleous 2002).

The Abbotsford-Sumas Aquifer (ASA) straddling the USA-Canada border is an ideal location to study landscape legacies on groundwater. Long-term monitoring (since the early 1970s) detected nitrate concentrations in exceedance of WHO standards (Wassenaar et al 2006, Mitchell et al 2003) and persistent elevated nitrate concentrations have remained a problem for decades. In recent decades, while the amount of total N applied over the aquifer has changed little, the source has changed substantially from inorganic fertilizer to manure. Furthermore, export of N from the region has declined leading to a surplus over the aquifer (Zebarth et al, 1998). The many complexities of this problem have challenged managers, farmers, and policy makers in both the USA and Canada who have initiated a wide variety of nutrient management strategies - with little apparent success - in reducing overall nitrate concentrations in the aquifer. This perceived lack of success may be in part a result of the time lags between the implementation of management practices and the residence time of nitrate in aquifers. Thus,

approaches to help understand water quality legacies across jurisdictional boundaries are particularly critical. Cross-border landscape indicators are particularly appealing in this context.

Within this context, I ask two questions to understand the historical dynamics of land cover change and its potential impact on nitrate concentrations in the ASA aquifer: 1) Are long-term groundwater nitrate concentrations changing over time and space? 2) What is the relative importance of historical versus contemporary LULC in explaining groundwater nitrate concentrations? To accomplish this, I examined trends in groundwater nitrate concentrations at monitoring wells across the aquifer. Then, I assessed temporal changes in a suite of landscape indicators using historical aerial imagery since the 1970s. I hypothesized that correlations between present day nitrate concentrations and historic LULC (1974 and 1996) may be important because of lag effects. Specifically, I hypothesized that present day nitrate concentrations would be more correlated with LULC from the 1970s-1990s than with contemporary LULC.

4.2 METHODS

4.2.1 Study Site

The Abbotsford-Sumas Aquifer (ASA) is a 200 km2 trans-boundary aquifer spanning the US-Canada border (Figure 4.1). The aquifer supplies drinking water for nearly 100,000 residents of Canada in the city of Abbotsford and the township of Langley as well as 10,000 people in the United States (towns of Sumas, Lynden, Ferndale, Everson, Nooksack, and scattered rural areas) (Chesnaux et. al. 2007). The unconfined, highly permeable sand and gravel aquifer lies within the agriculturally productive Fraser-Whatcom Valley and is recharged primarily by direct precipitation (Fraser Valley Soil Nutrient Study 2007). Over the last four decades, there has been a shift in land use from dairy production to more raspberry, blueberry, and poultry production.

Currently, red raspberry is the predominant agricultural crop in the region followed by significant forage grass and pasture (Zebarth 2015). Mean groundwater age is approximately 20 years and models predict an average of 6.3 years for water to travel by advection from the top of the water table to a position 10 m below the water table (Chesnaux, et. al. 2012).

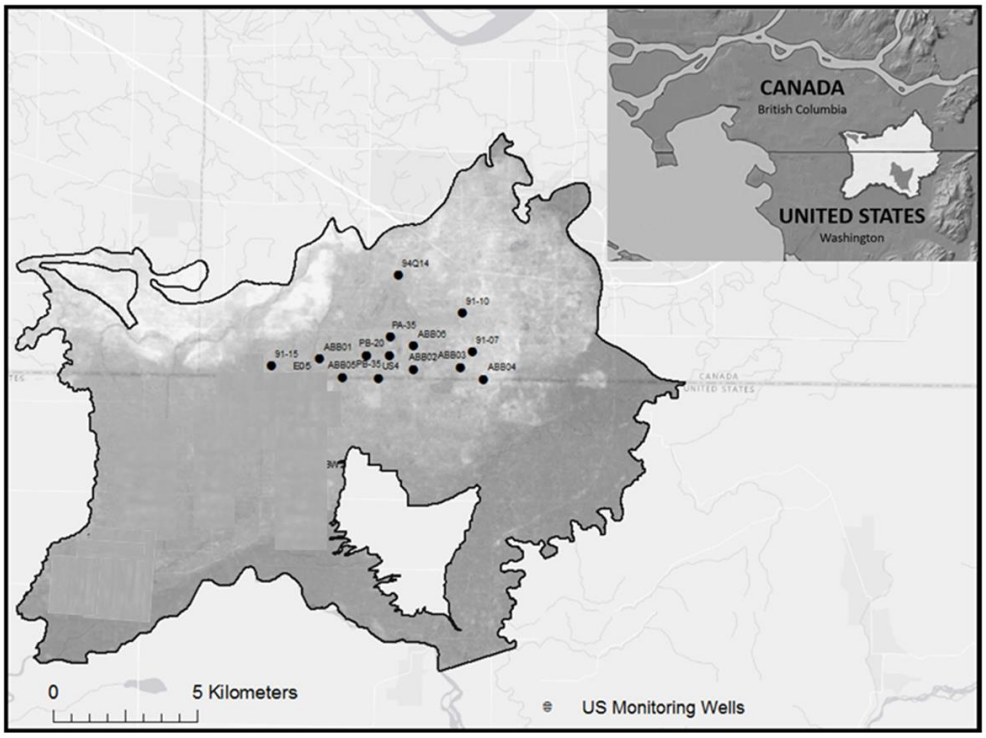

Figure 4.1. The Abbottsford-Sumas Aquifer (ASA) in southwestern British Columbia and northern Washington is a 200 km2 unconfined, highly permeable sand and gravel aquifer recharged primarily by direct precipitation. The aquifer has been the subject of longstanding nutrient management challenges affecting both countries.

4.2.2 Groundwater Nitrate Concentrations

I examined nitrate concentrations in 22 groundwater wells in Canada. Groundwater nitrate measurements [expressed in mg/L of nitrate as nitrogen (NO3-N)] collected from 1996-2016 were obtained from Environment and Climate Change Canada (ECCC) (Figure 4.2). Prior to June 2013, ECCC sampled monthly; however to reduce program costs, they began quarterly

sampling thereafter (i.e. March, June, September, December). Mean, median, minimum and maximum nitrate were determine on both a quarterly and an annual basis. To account for the depth of each well and its impact on the age of the water collected, I calculated the mean depth of mid-screen for each well (Figure 4.3).

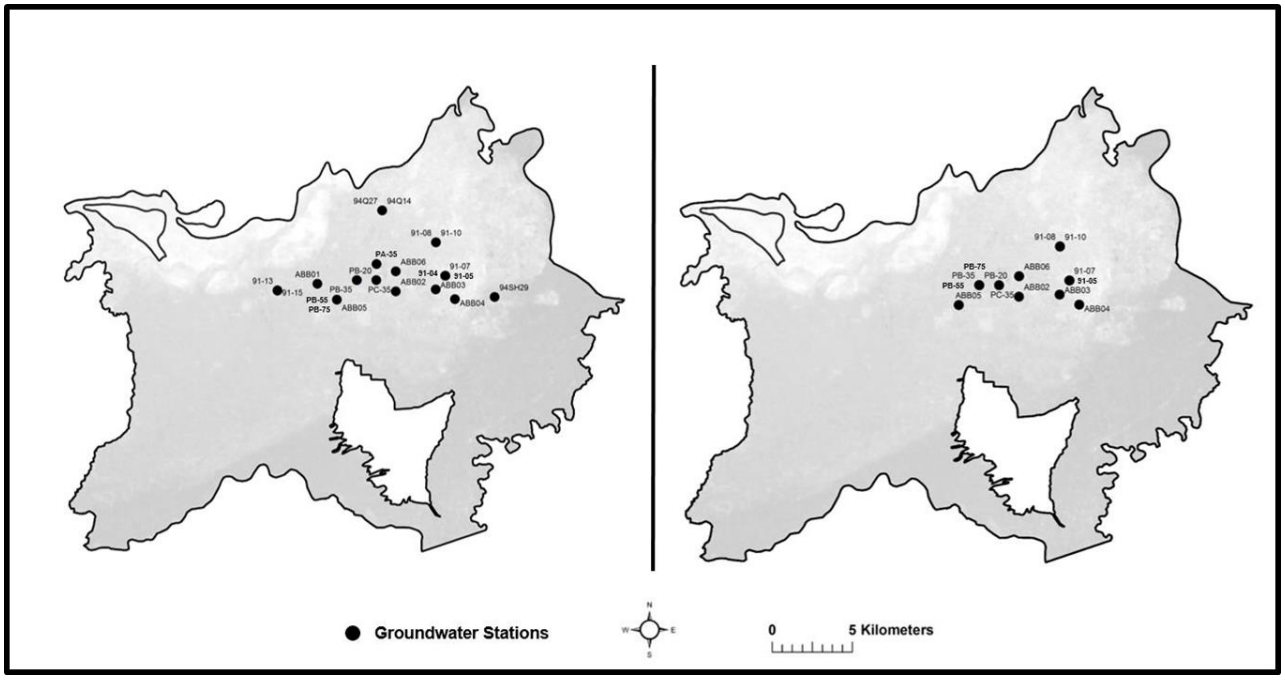

Figure 4.2. Groundwater monitoring stations in the ASA. A total of 22 wells were analyzed over time (1996-2016), while a subset of 14 wells were used to link land cover to nitrate concentrations.

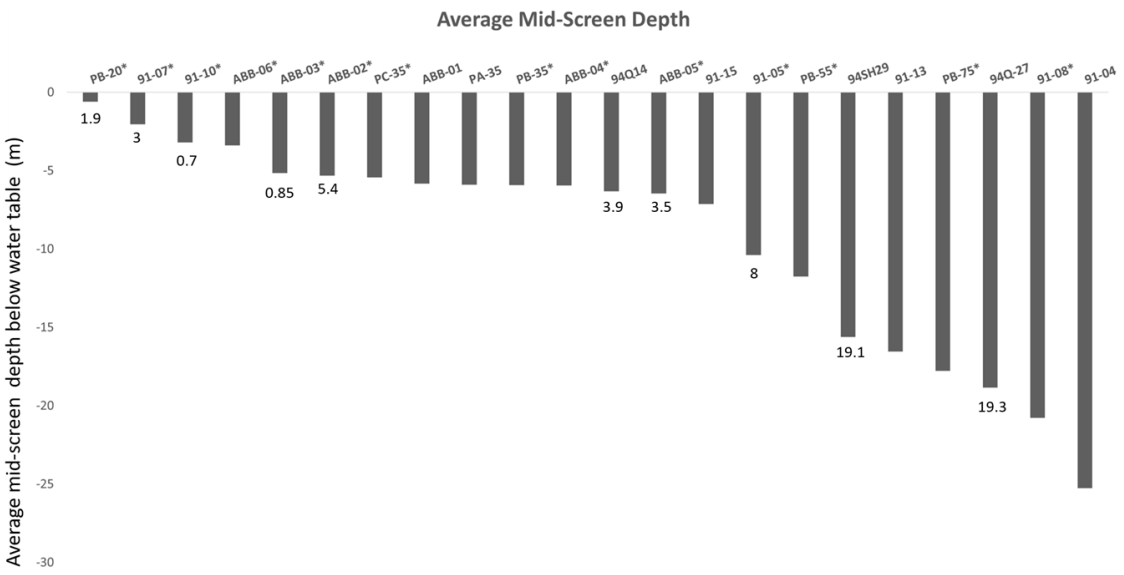

Figure 4.3. Mean depth of mid-screen below water table in meters (n=22). Wells with "*" indicate those used in land cover analysis (n=14). Numbers below the bars indicate age of water (years) based on H-He testing conducted in 2004 (Wassenaar, et al 2006) (n=10).

4.2.3 Geospatial Data

To characterize land cover change over time, I compiled and integrated a diverse suite of remotely sensed imagery, with special emphasis on characterizing agricultural features thought to influence nitrate concentrations. I collected historical aerial imagery from 1974 and 1996 from the National Air Photo Library (NAPL) from Natural Resources Canada, described next in more detail. I calculated contemporary (2012) land cover using a combination of Canadian Ministry of Agriculture ALUI data (2012), Metro Vancouver MVMS data (2010), and Metro Vancouver 5 m LiDAR-derived land cover (2014). As 2012 data were harmonized from >200 classes to 8 classes, per-class accuracy very likely increased (Aronoff 1982, Olofsson 2014). Table 4.1 further describes geospatial datasets (Gallagher and Gergel 2017).

Table 4.1. Comparison of characteristics of geospatial datasets used in this research. Concordance of ALUI and MVMS datasets is explained further in Chapter 3 Data Format: P = polygon, R= raster.

Dataset	Source	Year	Spatial Extent	Format	Spatial Resolution/ Minimum Mapping Unit	Number of Classes	Description of Classes
Agricultural Land Use Inventory (ALUI)	Canadian Ministry of Agriculture	2012	British Columbia	P	500 m^2 or width=10m	>200	Detailed info for ag (e.g., Anthropogenic > Terrestrial > Vegetated > Cultivated > Barley)
Metro Vancouver Multi-Spectral (MVMS)	Metro Vancouver Regional District	2010	Metro Vancouver Region	P	10 m	16	Urban classes (e.g., roads, built environments, urban mixed)
Aerial Photography	National Air Photo Library (NAPL) Natural Resources Canada	1974 1996	Photo-interpreted for limited farms throughout Greater ASA	R	1:10,000	---	See Table 4.2
Metro Vancouver Land Cover	Metro Vancouver	2014	Metro Vancouver Region	R	5 m	13	Used to derive hedgerow information for 2012 land cover

4.2.4 Aerial Photography

As I was interested in characterizing a suite of finer-scale features generally not visible using satellite imagery, interpretation of the aerial photographs was required. In order to do this, I trained and collaborated with an assistant to aid in the collection, georeferencing, photo-interpretation, and digitizing processes which took several months.

The photo acquisition process involved amassing hard copy aerial photographs (circa 1974) from Dr. Hans Schreier as well as downloading digital images from the National Air Photo Library (circa 1996). Color and grey scale aerial photographs captured in 1974 (1:5,000 scale) and 1996 (1:15,000) were scanned at high spatial resolution (1200 dpi) using a graphics grade scanner (Epson Expression 1640 XL). Images were then georeferenced and projected to WGS 1984 Mercator (auxiliary sphere) using ArcMap 10 (ESRI 2015). Ground control points were selected (approximately 8-10 per image) to minimize residuals (RMS error) and ensure no residuals exceeded 4.5 m. In total, geoprocessing and the subsequent digitizing of photos (explained next) took about two and a half months.

Within a 500 m radius of each groundwater monitoring station, we manually delineated land cover using six broad categories: Surface Water, Developed, Bare, Forest, Hedgerows and Cultivated lands using a minimum mapping unit (MMU) of 25 m^2 on 1974, 1996, and 2012 imagery (Table 4.2). Many days of training were required in order to ensure the interpretation of features was consistent between myself and my technician, with continuous verification of each other's work. When a polygon of 25 m^2 consisted of more than one land cover type, majority rule was used to assign the cover type to the dominant land cover class. It should be noted that there were a few instances in which we did not want to lose important class features (i.e. small hedgerows or small patches of trees that were clearly identifiable), therefore we digitized these

features even if they were smaller than our designated 25 m^2 MMU. I also delineated raspberry fields in 1974 and 2012 using ArcMap 10. Raspberry fields had previously been identified and delineated on the 1974 images and ground verified during the same decade by Dr. Hans Schreier and his colleagues. However, a complete lack of any source of independent historical ground verification information circa 1996 thwarted any attempt at rigorous identification of raspberry fields on 1996 images.

Table 4.2. Landscape indicators calculated within the 500 m wedge-shaped zone of influence surrounding each groundwater monitoring station using geodata described in Table 1. To consider the age of water collected at various well depths, mean depth of mid-screen was included in each model. * Not calculated for the year 1996 as raspberries were included in agriculture in 1996.

Landscape Indicator	Units	Description	Potential Relationship to Groundwater Nitrate
Agriculture	Proportion of Zone	Area used for the production of annual crops such as corn, soybeans, vegetables, as well as areas of grasses, legumes, or grass-legume mixtures planted for livestock grazing or the production of seed or hay crops. This class also includes all land being actively tilled.	Agriculture is main source of nitrate over the aquifer (Zebarth 1998, Chesnaux 2011).
Raspberries*	Proportion of Zone	Areas of raspberries fields (m^2) as indicated on aerial photos (only available for 1974 air photos)	Raspberries heavily fertilized in first 3 years of planting with declines in fertilizer application in subsequent years.
Vegetation	Proportion of Zone	Areas dominated by trees (deciduous, evergreen, and mixed)	Natural vegetation buffers impacts of nitrate contamination.
Bare Land	Proportion of Zone	Areas of bedrock, gravel pits, and other accumulations of earthen material.	Large gravel mining operations in the ASA may influence groundwater recharge rates. Impervious surfaces not included in this category but included in urban.
Developed Land	Proportion of Zone	Mixture of constructed materials and vegetation e.g., roads, lawns, barns, houses.	Anthropogenic activities increase impervious surfaces and decrease vegetation cover which can impact aquifer recharge rates.
Surface Water	Proportion of Zone	Open water, generally with less than 25% of vegetation or soil.	Streams and ponds may be surface-groundwater recharge zones and may lower nitrate levels.
Hedgerows	Proportion of Zone	Hedge or wild shrubs and trees, typically bordering a road or field. These do not include raspberries and blueberries but may include wild blackberries growing in the region.	Hedgerows absorb some nitrate running off fields thus reducing leaching through the soil into the groundwater.
Additional Co-Variates			
Depth of Mid-Screen	Meters	Depth of well mid-screen below the average water level. Average water level calculated from (1996-2016)	Shallow groundwater likely represents contemporary land use practices while deeper groundwater likely represents past land use.

4.2.5 Landscape Indicators

I aimed to use landscape indicators from different time periods to help understand the role of recent versus historical land use on groundwater nitrate concentrations. Thus, I quantified landscape indicators over multiple time steps within terrestrial zones of influence surrounding each well. Landscape indicators (Table 4.2) were quantified using three sets of imagery from 1974, 1996, and 2012. In addition, I calculated temporal change (Δ) in land cover between these three intervals: 1996-2012 (hereafter referred to as "recent Δ" land cover change), 1974-1996 ("historical Δ") and 1974-2012 ("long-term Δ"). Notably, due to limitations in aerial photo coverage among years, only a subset of 14 wells could be examined via this land cover change approach.

I further calculated landscape indicators within a wedge-shaped "upslope" zone of influence surrounding each monitoring well corresponding to the prevailing direction of lateral groundwater flow (Figure 4.4). Despite not addressing residence times or aquifer volume (which may be important in predicting nitrate concentrations), these zones helped spatially delimit potential contributing areas and incorporate known directions of flow. Landscape indicators within variable-sized radii (100 m, 500 m) were evaluated to identify which extent produced the strongest correlations between land use and nitrate concentrations. Due to a lack of statistical significance at some extents (i.e., 100 m) and other shapes (i.e., fully circular zones), I only reported results from models using a 500 m-sized wedge-shaped zone.

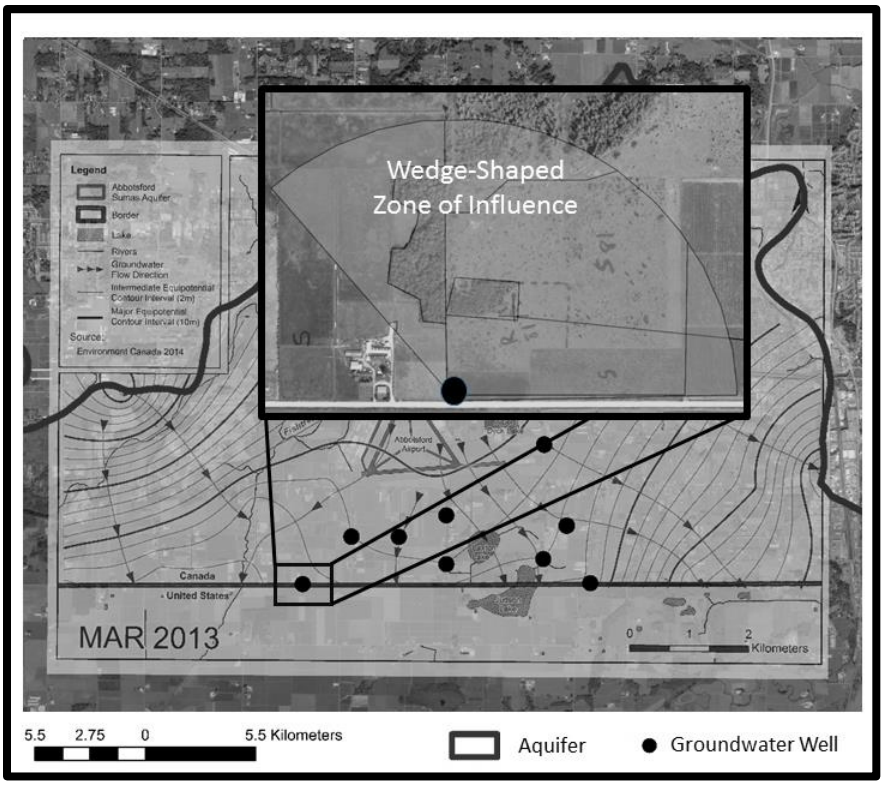

Figure 4.4. I characterized landscape indicators in terrestrial zones of influence within a 500 m distance surrounding wells and further used wedge-shaped zones to incorporate known direction of "upstream" groundwater flows.

4.2.6 Statistical Analysis

First, to examine long-term trends in nitrate concentrations, I performed Mann-Kendall (MK) tests to detect monotonic upward or downward trends over time (1996-2016). A monotonic upward (or downward) trend indicates a consistent increase (or decrease) through time, which may or may not be linear. A positive MK score indicates an increase with time whereas a negative MK score indicates the opposite. I then calculated Sen's slope estimates of the rate of change for these trends (Hirsch and Slack, 1984). Because the performance of MK tests is adversely affected when evaluating a time series with missing values, I interpolated missing values using means from two years before and after any missing quarterly samples.

Additionally, measurements collected in January 2007 were substituted for missing values in December 2006. Lastly, I mapped wells according to the significance and direction of trends.

My second approach examined connections between nitrate concentrations and landscape indicators. For nitrate measurements in 1996 and 2012, I determined measures of central tendency (mean, median) as well as minimum and maximum nitrate on both an annual and quarterly basis. I also used Sen's slope (derived from MK test from 1996-2012) as a dependent variable. I selected 1996 maximum nitrate, 2012 maximum nitrate, and Sen's Slope as dependent variables for further analysis as they showed the strongest correlations with landscape indicators in exploratory Pearson's correlation analysis.

To account for legacy nitrate and search for potential lagged effects of land use across the aquifer, I determined which landscape indicators best explained static (one year) and dynamic (Δ) measures of nitrate concentrations using concomitant static and dynamic landscape indicators. First, I determined if nitrate concentrations measured at a given point in time (1996 and 2012), were better explained by contemporaneous or prior (lagged) landscape indicators (1974, 1996, 2012) using backward stepwise regression. Second, I explored simultaneous changes (Δ) in nitrate and landscape indicators. To do so, I determined which periods of land cover change (Δ recent, historical, or long-term land cover change) were most significant in explaining changes in nitrate concentrations (represented by Sen's Slope).

My goal was to seek the best parsimonious models (with no more than 2-4 independent variables) and avoid model over-fitting as indicated by inflated R2 values and Akaike information criterion (AIC) scores. AIC aids model selection by evaluating the relative quality (or goodness-of-fit) of different statistical models by identifying and penalizing those which are over-fit (containing too many additional variables) (Burnham and Anderson 2002). Models with

the lowest AIC given a similar number of independent variables, indicate higher quality models. An additional model variable was warranted only if it lowered AIC by at least two points. I reported only significant models (p-value<0.1).

Shapiro-Wilk tests assessed the normality of independent and dependent variables and landscape indicators (e.g. proportion of raspberries, urban, etc.) were transformed as needed using the arcsine square root transformation. I used Pearson correlation coefficients to assess collinearity among independent variables and highly correlated (greater than r = 0.80) landscape indicators were not included in the same model.

4.3 RESULTS

4.3.1 Nitrate concentrations decreased in 30% of wells and increased in nearly 20%

MK tests were significant ($p < 0.10$) for eleven of the twenty-two wells. Seven of the eleven monitoring wells had decreasing trends in nitrate concentrations from 1996-2016 (Figure 4.5). Four wells demonstrated a significant increasing trend over the same period (Figure 4.6). The remaining eleven wells demonstrated no significant trends. **Figure 4.7** shows well locations noting increasing and decreasing trends. I also graphed boxplots showing the median, first and third quartile of nitrate levels for individual wells (Figure 4.8).

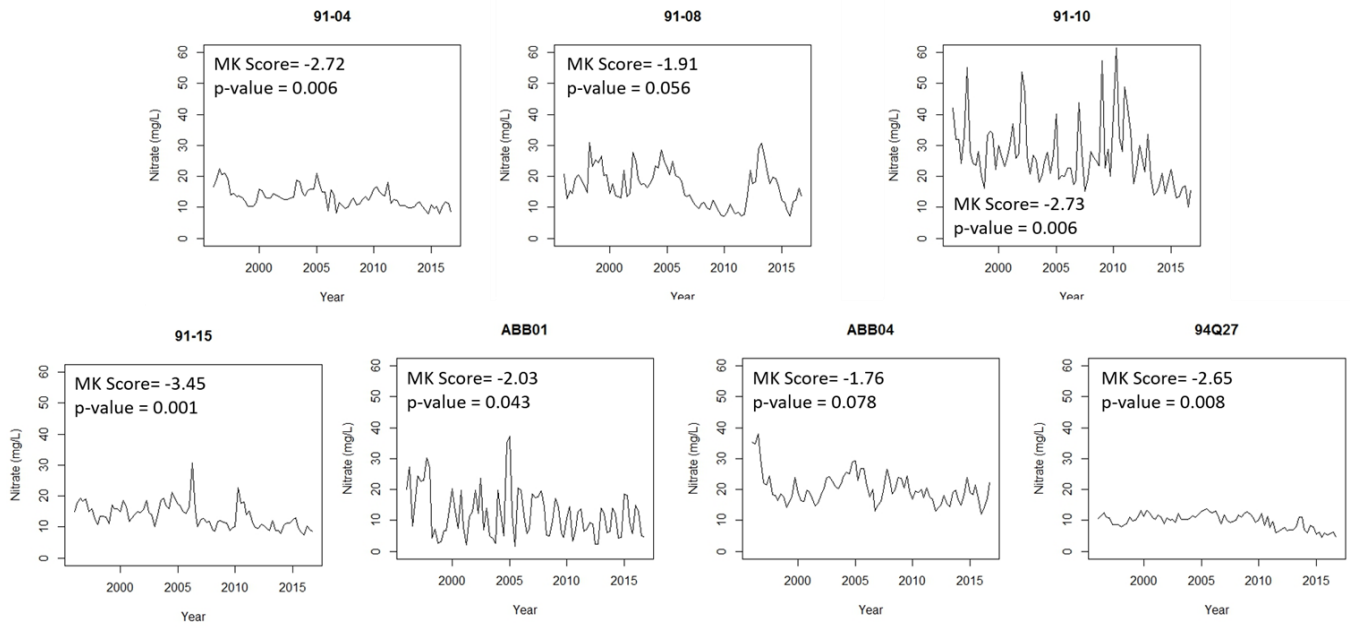

Figure 4.5. Seven of 22 wells exhibited declining nitrate concentrations over time (1996-2016) according to Mann-Kendall tests (p-value=0.10).

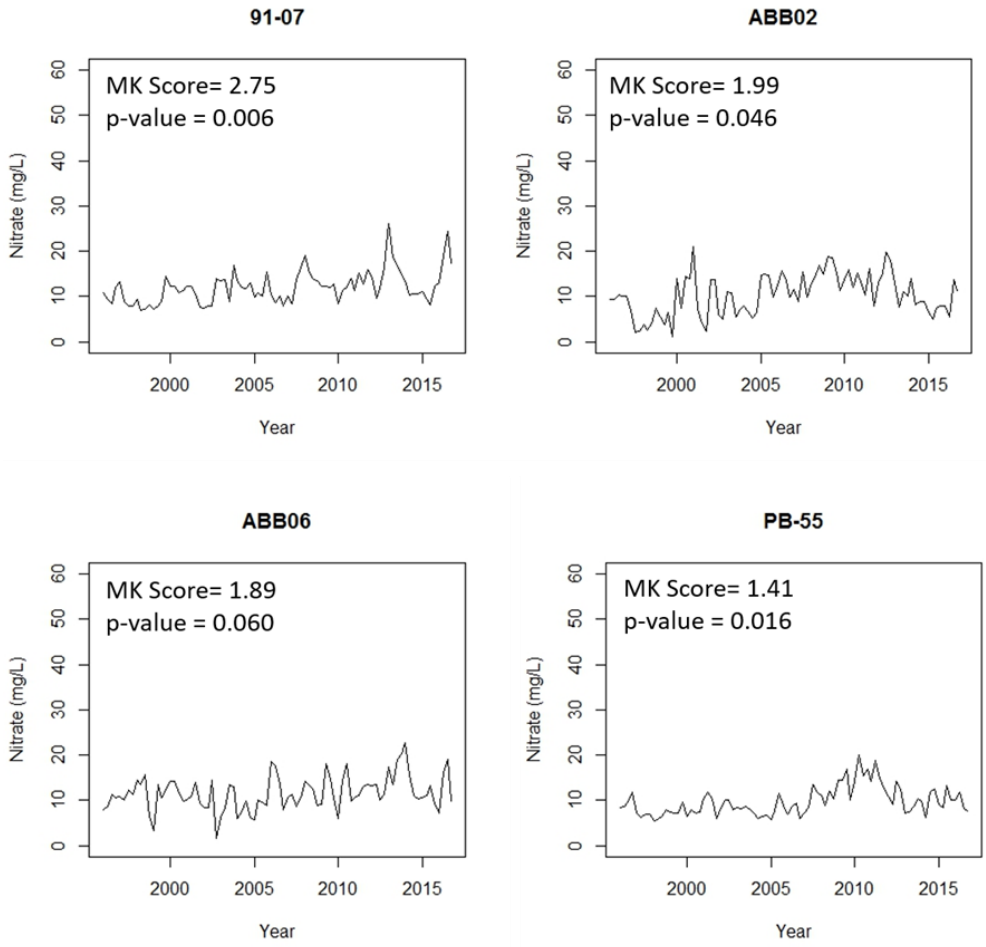

Figure 4.6. Four of 22 wells exhibited increasing nitrate concentrations over time (1996-2016) according to Mann-Kendall tests (p-value=0.10).

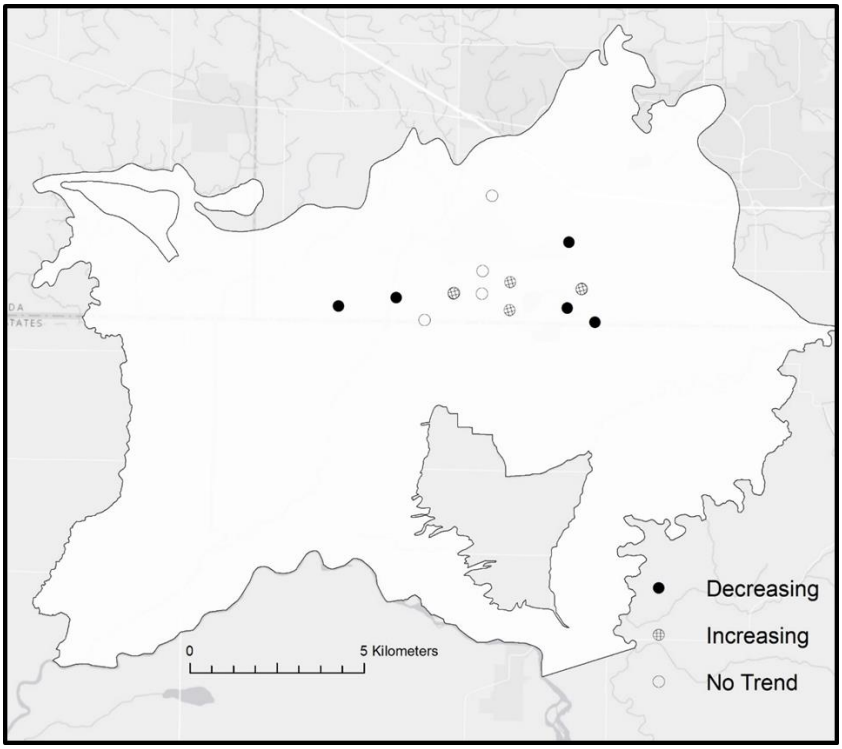

Figure 4.7. Direction of significant trends in nitrate concentrations (1996-2016) according to Mann-Kendall tests (p-value=0.10).

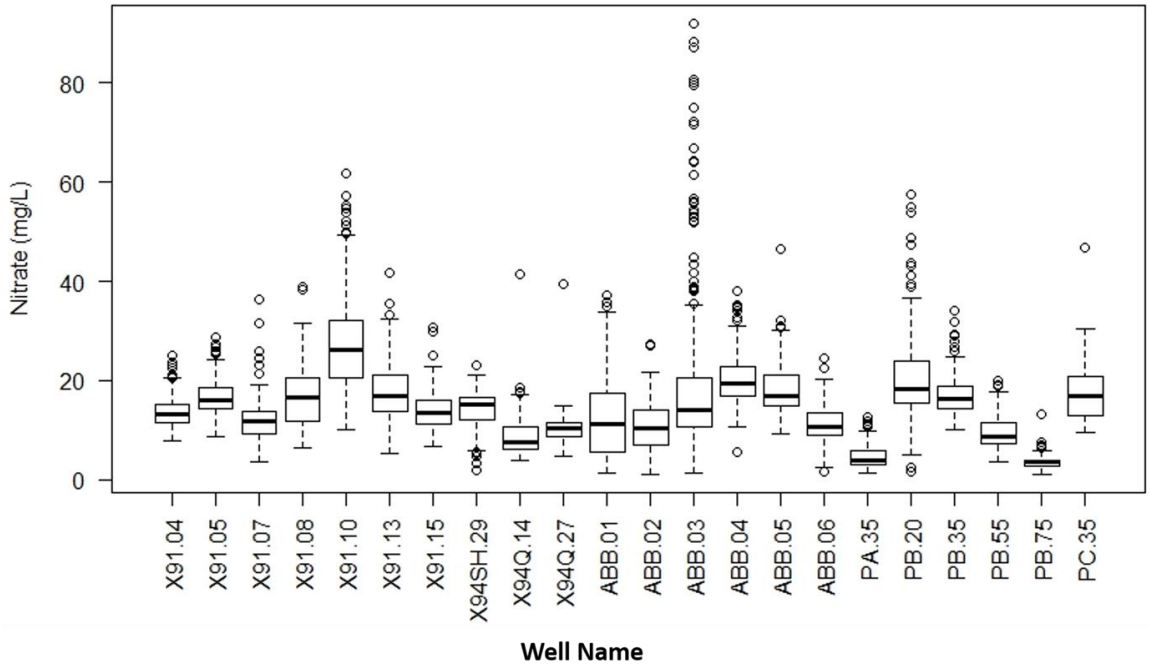

Figure 4.8. Box-plot displaying median N-concentrations, first and third quartile for individual wells (1996-2016) (n=22).

Next, I examined connections between water chemistry and landscape indicators. The strength of Pearson correlation coefficients indicated that Sen's Slope, 1996 maximum nitrate (hereafter referred to as 1996 nitrate) and 2012 maximum nitrate (hereafter referred to as 2012 nitrate) were most correlated to landscape indicators and thus deemed most suitable for further analysis.

4.3.2 Historical landscape indicators better explained nitrate than contemporary landscape indicators

Overall, the best models used lagged landscape indicators measured prior to the year of the nitrate measurement. I created eight models and the best model for each is in bold (Table 4.3). I further explain model results in subsequent sections.

The utility of models using historical indicators was made clear in an examination of nitrate concentrations from both 1996 and 2012 (Table 4.3A and 4.3B). Firstly, nitrate concentrations in 1996 were best explained by historical 1974 landscape indicators whereas land cover in 1996 did not explain any of the variance in 1996 nitrate (Table 4.3A). Land cover in 1974 was strongly correlated with 1996 nitrate (full model $R^2 = 0.71$, AIC=109). In this model, proportion of bare land was positively correlated with nitrate (partial $R^2 = 0.34$) while depth of mid-screen and hedgerows were negatively correlated with nitrate (partial $R^2 = 0.14$ and 0.11, respectively).

Table 4.3. We compared nitrate concentrations and trends to historic and contemporary land cover (1974, 1996, 2012) and landscape indicator Δ over time. * % Raspberries were calculated only in the years 1974 and 2012. To determine Δ in % Agriculture between 1974-1996 and 1996-2012, agriculture and raspberries were combined in each year 1974 and 2012 and then compared with 1996 land cover. The best model for each response variable is shown in bold.

(A) Past **(**1996) Maximum Nitrate, n=14

Year of Landscape Indicator	**Model R^2**	**Adj. R^2**	**p-value**	**AIC**	**Independent Variables** (and direction of relationship)	**Relative Importance** (Partial R^2)
1974	**0.71**	**0.58**	**0.01**	**109**	**+ Bare**	**0.34**
					- Depth of Mid-Screen	**0.14**
					- Hedge	**0.11**
					+ Raspberries	**0.06**
1996	N/A	N/A	N/A	N/A	N/A	N/A

(B) Contemporary (2012) Maximum Nitrate, n=14

Year of Landscape Indicator	**Model R^2**	**Adj. R^2**	**p-value**	**AIC**	**Independent Variables** (and direction of relationship)	**Relative Importance** (Partial R^2)
1974	**0.57**	**0.44**	**0.03**	**100.13**	**+ Raspberries**	**0.32**
					+ Vegetation	**0.15**
					- Depth of Mid-Screen	**0.09**
1996	**0.63**	**0.47**	**0.04**	**99.84**	**+ Vegetation**	**0.24**
					+ Agriculture (including raspberries)	**0.18**
					+ Developed	**0.12**
					- Surface Water	**0.08**
2012	N/A	N/A	N/A	N/A	N/A	N/A

(C) Sen's Slope (1996-2012), n=14

Timeframe of Landscape Indicator	**Model R^2**	**Adj. R^2**	**p-value**	**AIC**	**Landscape Indicator** (and direction of relationship)	**Relative Importance** (Partial R^2)
Long-term (1974-2012)	**0.83**	**0.73**	**0.005**	**-5.97**	**- Bare**	**0.35**
					+ Vegetation	**0.18**
					- Surface Water	**0.17**
					+ Agriculture	**0.07**
					+ Raspberries	**0.06**
Historic (1974-1996)	N/A	N/A	N/A	N/A	N/A	N/A
Recent (1996-2012)	0.47	0.31	0.08	6.24	+ Vegetation	0.23
					+ Developed	0.15
					+ Agriculture (including raspberries)	0.08

Secondly, 2012 nitrate concentrations were best explained by lagged landscape indicators from 1996 and 1974 (Table 4.3B). Historical land cover from 1974 and 1996 explained similarly substantive portions of the variance in 2012 nitrate (full model R^2 = 0.57, AIC = 100.13 and full model R^2 = 0.63, AIC = 99.84, respectively); whereas contemporaneous 2012 land cover was insignificant. Given the similar R^2 and AIC values for models with 1974 and 1996 land cover, no one best model was chosen and instead both models were deemed equivalent. Within each model, raspberries (partial R^2 = 0.32) and vegetation (partial R^2 = 0.15) from 1974 were positively correlated with 2012 nitrate, while depth of mid-screen was negatively correlated (partial R^2 = 0.09). Vegetation (partial R^2 = 0.24), agriculture (partial R^2 = 0.18), and developed land (partial R^2 = 0.12) in 1996 were positively correlated with 2012 nitrate.

4.3.3 Long-term land cover change best explained temporal trends in nitrate concentrations

Land cover change throughout the longest time frame available (1974-2012) most strongly explained the magnitude of Δ (i.e. Sen's slope) in nitrate concentrations (full model R^2 = 0.83; AIC = -5.96) (Table 4.3C). Bare land (partial R^2 = 0.35) and surface water (partial R^2 = 0.17) were inversely related to Sen's Slope indicating that as bare land and surface water increased, the rate of Δ in nitrate trends was dampened. This result does not necessarily indicate declining nitrate, but rather that the magnitude of change in nitrate trends (both positive and negative) diminished over time. In contrast, vegetation was positively related to nitrate trends (partial R^2 = 0.18). Notably, historic (1974-1996) land cover Δ was not significant in any model of Sen's slope; whereas recent (1996-2012) land cover Δ was moderately correlated (full model R^2 = 0.47, AIC = 3.29) with Sen's slope. In terms of recent land cover Δ, changes in vegetation

(partial R^2 = 0.23) and developed land (partial R^2 = 0.15) were positively correlated with Sen's Slope.

4.4 DISCUSSION

4.4.1 Past land cover explains contemporary groundwater nitrate concentrations

The results support the notion that past land use can have lasting impacts on groundwater. For my first objective, I sought to understand if long-term nitrate concentration were changing over time and space. MK results showed nitrate increased in 22% of wells from 1996-2016, yet decreased in 31% of wells. For my second objective, I sought to understand the relative importance of historical versus contemporary LULC in explaining groundwater nitrate concentrations. Nitrate concentrations were strongly correlated with lagged land cover yet rarely with contemporary land cover. The strongest models of nitrate concentrations consistently suggest that bare land, raspberries, depth of mid-screen and vegetation explain as much as 83% of nitrate trends over time and 63% of nitrate concentrations measured in a given year. In models predicting Sen's slope (i.e. magnitude) of nitrate trends over time, vegetation and developed land (i.e. built areas and impervious surfaces) were positively related whereas bare land and water were inversely related. For models using 1996 and 2012 nitrate as a response variable, bare land and raspberries demonstrated consistent positive correlations. Additionally, vegetation, agriculture, and developed land were consistently positively correlated with 2012 nitrate. Hedgerows were inverse related to 1996 nitrate while depth of mid-screen and surface water coverage was negatively correlated with 2012 nitrate.

4.4.2 Heterogeneous nitrate trends suggests heterogeneous patterns of nitrate application

The nitrate increases seen in some wells coupled with declines in others, suggests several possibilities including: 1) highly localized nitrate contamination and/or 2) spatial autocorrelation

among wells. The fact that many of the wells with decreasing nitrate are located in close proximity to one another may mean that there are “hotspots” of nitrate in soils and therefore well measurements are not independent of one another. These hotspots may be related to individual farm management strategies as soil types across the ASA are quite homogenous. Thus, variation in fertilizer applications and irrigation at the field-scale may explain differences in nitrate trends across the aquifer. Chapter 3 also suggested the important role of raspberry field renovations, pointing to the impact of soil turnover and organic fertilizer amendments on nitrate concentrations in the ASA (Gallagher and Gergel 2017).

Hedgerows, buffer strips, cover crops, and nutrient management plans are effective best management practices (BMPs) proven to reduce nitrogen runoff to surface waters (Garcia-Diaz et al 2017, Blanco-Canqui et al 2015). Although I could not account for the impact of nutrient management plans on individual farms, this would be an important next step and could help to explain heterogeneous trends in nitrate across the aquifer. The nitrogen cycle is a complex biogeochemical cycle influenced by both biological and physical processes. Thus, generalization about the state of nitrate within an aquifer should be made with great caution. My heterogeneous results further bolster the need for caution in extrapolation of aquifer conditions throughout the larger area based on this sample of wells.

4.4.3 Lagged land cover impacts are an important consideration for landscape management

While BMPs could potentially reduce groundwater contamination, beneficial results of landscape management could take years or even decades to materialize as leaching rates, atmospheric conditions, and water residency times affect measured aquifer contamination (Mulla et. al. 2008). Benefits could take longer to materialize in groundwater systems due to substantive

lag times between fertilizer application on the ground surface and arrival in the water table. Leaching rates can be influenced by geological factors (i.e. rock type, pores space size) as well as atmospheric conditions (i.e. greater precipitation rates).Furthermore, it may also take years or decades for initial contamination to even be detected. Additionally, based on the ASA's average groundwater flow rate of 30 m y^{-1}, increasing the zone of influence may help to better explain legacy effects of land cover on groundwater quality (Hii et al., 1999) as a larger zone may capture additional potential N sources from earlier years due to the amount of time it takes for contaminates to move laterally through rock material within the aquifer. Furthermore, as per the work of Chapter 3, the inclusion of soil data was not warranted in the models. However, soil organic matter estimates would be a helpful long-term indicator as this would help explain the dynamic nature of agricultural soils. The lack of annual land use and soil data may lead to mismatched legacies as the history of each field is vital to understanding and creating proxies for residual a field's soil nitrogen. Though yearly historical data is not readily available, moving forward records of land use and crop types would be beneficial for understanding potential nitrate leaching.

4.4.4 Historical air photos have a broad application in exploring landscape legacies of groundwater

Though the importance of landscape history on contemporary ecosystems has become increasingly apparent (Rhemtulla and Mladenoff 2007, Tomscha and Gergel 2016, Tomscha and Gergel 2015), relatively few studies examine water quality legacies using historical land cover from aerial photography. Many have used aerial photography to explore long-term changes in riverine systems (Large and Petts 1996, O'Connor et. al. 2003, Tomlinson et. al. 2011, Wan et. al. 2014) and even ecosystem services (Tomscha and Gergel 2016), very few have used historical

aerial photography to examine groundwater quality (Swartz et al 2003). My approach is transferable to other regions (which likely have aerial photography from the 1930s or 1950s) (Morgan et. al 2010, Morgan and Gergel 2013) and could be useful in evaluating other non-point source pollutants such as phosphorus, arsenic, and heavy metal contamination within aquifers.

4.4.5 A lack of nitrogen information is complicating management

Accurate quantification of nitrate leaching to groundwater is challenging due to the complex interaction between land use practices, on-ground nitrogen loading, groundwater recharge, soil nitrogen dynamics, and soil characteristics. Several models and frameworks estimate nitrate contamination of groundwater using parameters such as soil type, land use and land cover type as proxies for nitrogen loading (Almasri 2007, Narula and Gosain 2013, Bernardo et. al. 1993). Specific nitrogen loading values for my landscape indicators are difficult to determine, likely to be highly variable by region, but should be prioritized in future work. That said, my results are consistent with SPARROW (Spatially Referenced Regression on Watershed attributes) (Wise and Johnson 2013), a model which attributes N sources, when applied to Pacific Northwest surface water catchments. Despite their utility in many data-limited situations, weakness are apparent when validating model predictions of nitrate leaching (Fox et. al. 2001), however. This work shows that landscape indicators are a useful tool for creating proxies for N-loading and evaluating potential sources and can supplement and help fill this critical gap.

4.5 CONCLUSIONS

Non-point source (NPS) pollutants, such as nitrogen, are recognized as the single greatest threat to surface and subsurface drinking water (Loague et. al 1998). Nutrient runoff from agriculture is issue of global importance (Tilman et al 2001) and historical land-use can have lasting impacts on present-day water quality (Harding et. al. 1998). Despite impacting much of

the Earth's land surface, data on non-point source pollutants such as nitrate are not readily available in many regions (Dressing et al 2016). A lack of accurate information hobbles analysis thus complicating aquifer management in many regions (Gleick and Cain 2004, Henriksen et.al. 2007).

Within the aquifer, I found nitrate concentrations increased in nearly 1/4 and decreased in nearly 1/3 of wells from 1996-2016. Such inconsistent trends in nitrate concentrations across the aquifer may indicate temporal variability in N loading as well as heterogeneous time lags operating. I also found contemporary nitrate concentrations were more strongly correlated with historic than contemporary land cover. This result may indicate longer residence times for nitrate in the aquifer than previously assumed with important implications for nutrient management strategies. Moving forward, management of contemporary landscapes will in turn have repercussions for future water quality (Bennett et. al. 2001; Van Meter et. al. 2016). As such, understanding the persistent legacy "echo" effects of land-use on aquatic systems is essential to create policies to improve water quality.

Chapter 5: Conclusions

Nearly 600 transboundary aquifers worldwide provide drinking water and irrigation to millions, contributing to human health and economic development (IGRAC, 2015). Urbanization and agricultural intensification have contributed considerably to contamination of these groundwater resources as well as many cross-border lakes and watersheds. There are ten trans-boundary aquifers shared between the USA and Canada and numerous more surface water bodies (Rivera, 2015). Many of these shared aquatic resources, such as the Great Lakes basin, St. Lawrence, as well as the Nooksack, Souris, Milk and Mary river systems face challenges not unlike those facing the ASA region. The International Joint Commission, a bi-national organization established by the governments of the United States and Canada under the Boundary Waters Treaty of 1909, has provided a mechanism for cooperative management of the St. Lawrence Great Lakes and other cross-border waters. Although much has been accomplished in managing surface waters across borders, few policy measures have been implemented to manage shared groundwater resources. Furthermore, managing cross-border groundwater resources has proved challenging as spatial and temporal data collected by different countries are often not compatible nor shared among jurisdictions.

The approaches used to monitor and assess land cover in this dissertation are affordable and easily transportable to transboundary water systems across the US-Canada border and beyond. The overall objective of this dissertation was to develop a framework for understanding linkages between land use and land cover (LULC) and nitrate trends within the context of transboundary aquifers. I developed several innovative spatial approaches and pushed the temporal boundary of aquifer evaluation via integration of multi-method approaches for long-term assessment. A number of key findings emerged from three primary data chapters. I

conclude by describing these key findings and discuss the limitations of my approach. I also suggest directions for future research which include using future scenario planning to engage communities and combat groundwater contamination from a socio-ecological perspective.

In **Chapter 2,** I explored urban-rural gradients in order to spatially and quantitatively describe long-term landscape changes surrounding groundwater dependent cities from 1990-2015. I quantified landscape composition and configuration for eleven cities in the US and Canada and compared these patterns among cities and across borders. I found that evenness among land cover types increased in all cities indicating that proportional abundances of land cover types became more similar. I determined this was driven by an increase in urban land while forest and agricultural land declined. Additionally, I found greater forest loss in Canada but greater losses of agricultural lands in the USA. This difference in land conversion could be a direct link to differences in land use policies north and south of the border. This chapter provided important regional context for the analyses tackled in Chapters 3 and 4.

In **Chapter 3** I highlighted the use of high spatial resolution imagery to assess fine-scale landscape features mechanistically linked to nitrate loading in a transboundary aquifer. I found that proportions of different berry types, forage/pasture, and area of fields undergoing renovations consistently explained measured groundwater nitrate concentrations. The strongest models explained as much as 72% of nitrate concentrations. By incorporating areas of raspberry renovations (i.e., field replanting), landscape indicators were able to capture potential mechanisms associated with land use practices, not just land use or land cover. As surprisingly few studies have quantitatively linked groundwater nitrate concentrations to land use, land cover, or land use practices, my research provides an important new approach that is transportable to other regions facing similar challenges.

In **Chapter 4** I explored long-term landscape legacies likely linked to nitrogen pollution by using historical landscape indicators. Building off the general approach and key results of Chapter 3, I further evaluated longer-term trends in nitrate concentrations over 10 additional years. To account for potential lagged impacts of prior land cover, I also quantified a suite of historical landscape indicators depicting nitrate sources over several decades prior while also taking into account the direction of groundwater flow within the vicinity of each well. While many studies use Landsat imagery to derive land cover, I used high spatial resolution historical aerial imagery in order to map fine scale features mechanistically linked to nitrate loading, such as hedgerows and raspberries. I found that contemporary nitrate concentrations, as well as long-term trends in nitrate, were much better explained by historical than contemporary landscape indicators. Raspberries, bare land, and vegetation consistently explained 63-83% of model variance in nitrate. As excess groundwater nitrate contamination from agricultural intensification continues to be a global concern, this work helps develop affordable and repeatable monitoring approaches for understanding the persistent "legacy" effects of land use on groundwater.

5.1 Caveats and considerations

This dissertation presents a rare highly detailed construction of cross-border LULC to create landscape indicators useful in tracking change over multiple decades. However, using cross-border and historical data for such purposes is not without limitations. One limitation, the lack of consistent transboundary data sources, is not merely a problem distinct to the Greater ASA region, but is a global challenge. Often, too little attention is given to the quality and content of data (Verburg et al, 2011) yet is critical to recognize that different sources of data will have different strengths and weaknesses for any particular application. Comparison and integration of different data sources is often hampered by a myriad of issues including:

1. Temporal consistency: e.g. repeated natural resource inventories rarely employ the same methods as previous studies due to changing technology, science and policy objectives (Wadsworth et al, 2008). This is challenging to address as it is often impossible to successful resample and recreate most datasets. To address this limitation in my own study, however, I was able to utilize a long-term nitrate dataset collected and overseen by one agency. Though the sampling frequency changed throughout the course of monitoring, I subsampled the data to bolster consistency of comparisons made over time.

2. Spatial consistency and scaling bias: i.e. valid comparisons between datasets cannot be made unless the same spatial relationships can be assumed for all areas being compared (Williams et al, 2002); a universal problem in land use planning and decision-making. To address this issue, I compared among spatial datasets of similar spatial resolution (e.g. 30 m resolution in Chap 2). And where data sets were not of the same scale I reclassified datasets and harmonized data.

3. Thematic differences and inconsistencies in data sources: a range of differing terms and definitions can be used to identify (essentially) the same land cover class in different LULC inventories. For example, official definitions of forest cover may include a wide range of canopy cover between 10 and 80% (Wadworth et al 2008). Thematic differences and inconsistencies are a non-trivial issue in harmonizing cross-border geospatial datasets. Aggregating several hundred incongruous classes from two different jurisdictions to just a few classes presents an array of challenges. In order to minimize thematic differences, I used ArcGIS to aggregate, reclassify, and thus harmonize imagery to the specific classes needed for the scope of my work. Despite significant attention to

detail in this process, some classes may be inconsistent because of uncertainties in the original classification schemes. However, each data source used in Chapter 2 and 3 had a minimum 85% class accuracy rating from the agencies providing the data. Furthermore, I reclassified datasets and grouped classes, in some cases from several hundreds to less than ten, thus increasing map accuracy (Olofsson et al 2014).

4. Differences between land cover and land use: the distinction between land *use* and land *cover* is an important issue underlying differences in how land is classified. The relationship between land cover and land use is one of the major challenges for monitoring, modelling, and communicating land change (Comber 2008, Verburg et al 2009). To address this, I created my own landscape indicators which incorporated land cover, land use, and specific land use practices relevant to my specific environmental questions of interest.

In addition to challenges of transboundary land cover data, the use of historical land cover data adds another set of challenges including confirming accuracy of the images being used. One way to confirm accuracy of digitized historical aerial images is to compare the images to other historical maps. Another method of ensuring accuracy of data, is to discuss land use and landscape changes with older and long-time residence within the study area. Finding additional historical images and interviewing long-term residents can be a difficult and time consuming process, however, this can greatly improve confidence in the datasets being used (Aronoff 1982).

5.2 Future research directions and applications: Future scenario planning

A major challenge for agriculture in the 21st century is producing adequate amounts of food while simultaneously protecting environmental quality along with the health and livelihoods of rural communities (Sutton et al 2013). This challenge is especially apparent throughout the ASA region, the undisputed raspberry capital of North America. With its abundant rainfall, fertile soils, and prime climatic conditions, Whatcom County, WA, USA produces more than 65% of US' red raspberries, while nearly 98% of Canada's raspberries are produced in Abbotsford, BC (Lynden Chamber of Commerce 2015, Ministry of Agriculture 2015). Additionally, the region has now also started producing blueberries over the last 20 years. As described in previous chapters, agriculture in the ASA has undergone a deep process of change in recent decades. Reforms to government policies, the volatility of food prices, and the emergence of new driving forces will continue to shape the future of farming activities in the region. In tandem, nitrate contamination remains an on-going issue throughout the aquifer with concentrations in many wells exceeding the legal limit of 10 mg/L-$N0_3^-$-N (Chesnaux et al 2007). Decades after widespread nitrate contamination was first identified in the aquifer, elevated nitrate concentrations remains a persistent trans-boundary water quality concern in the USA and Canada. Envisioning a positive future for agriculture and water quality requires innovative thinking.

While the data used in my dissertation was limited in scope to historical and contemporary time periods, additional research should develop future scenarios as a way to explore even more creative options for managing agricultural-based sources of nitrogen. Scenarios are not predictions, forecasts, projections, nor recommendations, but rather are intended to be imaginative "what if" situations that help people visualize different but plausible

future pathways (Raskin et al 2005). Scenario planning could help to open this discussion and engage local farmers in envisioning and considering plans for positive futures in the region. Scenarios are extremely well-suited to exploring situations where uncertainty is high and controllability is low (Peterson et al 2003). Scenarios can also help policy makers decide what needs to happen today in order to achieve goals in an uncertain future.

I found that over the last 25 years, urban land increased on average 13% while agriculture and forest cover decreased 9% and 12%, respectively. Therefore, creating scenarios in which these rates stay the same, double, or decline can help to predict the consequences of such LULC changes on groundwater quality and help formulate adequate responses and land use planning strategies. Furthermore, combining these changing LULC scenarios with existing groundwater models such as USGS's MODFLOW, Pacific Northwest Laboratory's STOMP, and the University of Michigan's Interactive Ground Water (IGW) can help to better understand how future land use changes will impact groundwater nitrate concentrations. Consideration of several contrasting future scenarios can help policy-makers design plans that seek options that are robust to many futures. Thus, linking scenarios to groundwater models can facilitate transition processes (Jarchow et al 2012). Future scenario work should not only include land cover change results for planners and managers, but also present and explore scenarios in several different ways for various audiences, such as visuals and narratives, to better communicating with the public and regional farmers.

In addition to scenarios, other innovative ideas for managing and monitoring water quality in the region includes consistent land use and land cover mapping using imagery collected from drones, tower-based instruments, Lidar, RapidEye, or Sentinel-2 (Wulder et al 2004, Kussul et al 2016). These approaches would yield higher spatial and temporal information

regarding drivers and thus create opportunities to identify additional landscape indicators (such as barn size, barn type, crop density, and annual/seasonal changes in crop cover). Though this data is rarely available for historical land cover studies, moving forward, if collected seasonally over many years, this data could aid understanding of long-term soil dynamics for individual fields and be used to more accurately estimate residual soil nitrogen for an individual field. Utilizing such data could help to create better models for correlating land use and land cover with water quality data.

5.3 In Summary

As landscapes transition to more urban and agriculturally-intensive uses at rapid rates, understanding landscape interactions with surface- and groundwater systems through better landscape indicators is increasingly necessary. Collecting information across privately-owned agricultural landscapes, especially those that span political boundaries, can present many challenges ranging from expensive sampling costs, unequal sample sizes, and spatio-temporal gaps in monitoring. Landscape pattern metrics and landscape indicators derived from remote sensing provide affordable, transportable and relatively simple approaches to classifying and quantifying landscapes. Such tools, when employed consistently across large areas over long time periods, can potentially aid in better understanding the transitions occurring in trans-boundary landscapes and support implementation of effective and holistic management practices. Determining historical legacy impacts of past land use will help to better understand how contemporary land use and policies might affect future nitrate concentrations in aquifers. As such, the long-term approaches explored in this dissertation lay the foundation for further understanding and managing of groundwater resources.

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Appendices

Appendix A

Land Cover	City	PLAND (%)	City	PLAND (%)	City	PLAND (%)
Un	Abbotsford	0.00	Chilliwack	0.00	Lynden	0.01
Ot		2.81		4.32		0.66
Cr		40.22		31.20		53.54
Fo		20.14		35.48		8.19
Ur		30.80		19.69		12.42
Veg		3.19		4.03		20.78
Wa		2.44		4.57		0.95
We		0.41		0.71		3.45
Un	Anacortes	6.88	Hope	0.00	Maple Ridge	0.00
Ot		1.85		1.11		0.59
Cr		0.95		0.23		10.25
Fo		52.66		86.55		44.50
Ur		24.91		2.22		36.86
Veg		7.59		7.06		2.25
Wa		2.57		2.06		4.63
We		2.58		0.76		0.92
Un	Blaine	1.77	Kent	0.00	Mount Vernon	0.53
Ot		0.31		4.81		34.27
Cr		0.72		18.93		22.48
Fo		42.23		50.42		22.20
Ur		34.82		7.67		14.00
Veg		13.60		8.25		2.72
Wa		3.85		8.72		3.80
We		2.69		1.21		0.53
Un	Bellingham	10.34	Langley	0.00		
Ot		1.28		0.57		
Cr		0.38		23.31		
Fo		32.14		25.30		
Ur		24.03		45.34		
Veg		17.71		3.03		
Wa		0.31		2.10		
We		13.81		0.36		

Table A.1. 2015 percent land cover for each class for each city.

Land Cover	City	PLAND (%)					City	PLAND (%)					City	PLAND (%)				
		0-2	2-4	2-6	6-8	8-10		0-2	2-4	4-6	6-8	8-10		0-2	2-4	4-6	6-8	8-10
Un	Abbotsford	0.00	0.00	0.00	0.00	0.00	Chilliwack	0.00	0.00	0.00	0.00	0.00	Lynden	0.00	0.00	-0.04	-0.04	-0.02
Ot		4.11	1.83	3.62	1.83	2.34		1.66	2.59	2.84	5.23	4.59		0.94	0.41	0.28	0.35	0.58
Cr		4.93	0.60	-5.00	2.39	3.37		1.33	0.63	-2.64	-1.11	-0.87		-38.30	-17.85	-22.53	-26.36	-27.44
Fo		3.29	-5.24	-8.43	-9.78	-11.12		-1.23	-6.12	-5.46	-6.73	-5.36		-8.12	-3.76	-2.18	-5.28	-10.61
Ur		-13.28	1.79	8.92	4.73	3.01		0.37	3.81	5.94	5.32	3.05		39.68	11.30	6.39	6.12	8.06
Veg		0.45	0.87	0.74	1.74	2.87		-2.22	-0.31	1.15	4.25	3.82		4.88	9.24	16.39	22.15	24.50
Wa		-0.03	-0.06	-0.02	-0.65	-0.79		0.00	-0.76	-2.12	-7.81	-6.01		-0.59	-0.19	-0.32	-0.12	0.49
We		0.52	0.21	0.16	-0.25	0.33		0.10	0.16	0.29	0.84	0.78		1.52	0.85	2.01	3.17	4.45
Un	Anacortes	0.99	4.03	5.73	7.96	8.15	Hope	0.00	0.00	0.00	0.00	0.00	Maple Ridge	0.00	0.00	0.00	0.00	0.00
Ot		-0.18	1.86	-0.25	0.71	-0.81		8.38	2.03	0.62	0.48	0.67		0.16	0.45	0.70	0.50	0.53
Cr		-1.64	0.14	-10.19	-6.33	-1.24		0.00	0.04	-0.01	-0.01	-0.01		0.07	-0.47	0.09	-7.29	-8.63
Fo		-25.68	-16.71	-20.15	-23.82	-8.68		5.51	-2.42	-5.07	-6.53	-7.99		-21.55	-17.91	-14.78	-10.01	-10.09
Ur		27.52	15.41	13.99	11.94	5.74		-16.10	-3.61	-1.06	-0.53	-0.47		21.35	18.23	14.73	16.64	17.52
Veg		-1.27	-2.39	10.00	7.83	0.55		8.45	6.09	5.85	6.80	7.83		0.28	1.06	2.40	1.01	1.09
Wa		-0.08	-3.55	-1.78	-1.65	-5.78		-8.09	-3.44	-0.87	-0.73	-0.63		-0.28	-1.02	-0.61	-0.70	-0.62
We		0.35	1.21	2.66	3.37	2.06		1.85	1.30	0.55	0.51	0.60		-0.03	-0.34	-2.53	-0.15	0.20
Un	Blaine	29.03	34.78	5.23	0.21	0.23	Kent	0.00	0.00	0.00	0.00	0.00	Mount Vernon	0.00	0.00	0.00	0.00	0.00
Ot		-0.06	0.56	-0.48	-0.04	0.39		1.39	6.27	6.74	4.24	3.72		0.04	0.41	0.09	0.18	-1.56
Cr		-14.91	-21.14	-27.48	-26.30	-34.51		-8.49	-5.60	-1.56	-2.68	-2.57		-12.53	-11.03	-12.21	-10.78	-3.98
Fo		-15.01	-12.70	-16.04	-16.64	-20.00		-8.46	-7.00	-7.03	-4.97	-3.32		-21.95	-23.35	-16.38	-12.57	-13.15
Ur		24.94	12.47	21.32	15.02	11.24		0.57	-0.78	1.77	2.28	1.70		38.45	25.84	13.98	10.34	8.92
Veg		-0.44	9.44	10.49	14.64	22.80		14.95	13.11	8.38	6.40	5.14		-4.14	6.88	11.62	9.91	8.50
Wa		-27.29	-32.89	-5.04	-0.12	0.04		-0.01	-6.47	-9.44	-6.25	-6.09		-0.26	-2.53	-0.92	-1.06	-0.48
We		3.72	9.48	12.00	13.23	19.81		0.05	0.47	1.14	0.98	1.43		0.39	3.77	3.81	3.98	1.75
Un	Bellingham	2.70	1.35	0.17	1.11	2.99	Langley	0.00	0.00	0.00	0.00	0.00						
Ot		0.14	-1.02	-0.83	-0.98	-1.14		0.19	0.15	0.34	0.29	0.79						
Cr		0.02	-1.28	-3.91	-5.87	-13.61		-7.84	-10.05	-2.26	-0.93	0.07						
Fo		-13.95	-23.45	-24.34	-21.26	-23.19		-17.29	-11.76	-13.74	-18.26	-19.87						
Ur		19.86	29.65	23.87	13.87	11.64		23.27	20.41	14.14	17.32	17.58						
Veg		-6.79	-4.33	4.72	12.00	18.87		1.54	1.25	1.59	1.71	1.65						
Wa		-2.08	-0.93	-0.17	-0.27	-0.61		0.02	-0.03	-0.04	-0.21	-0.52						
We		0.10	0.02	0.49	1.39	5.05		0.12	0.02	-0.04	0.07	0.30						

Table A.2. Changes in land cover types for each city along a rural to urban gradient (0-2 km, 2-4 km, 4-6 km, 6-8 km, 8-10 km radius).Canadian cities are highlighted in red and US cities are highlighted in blue

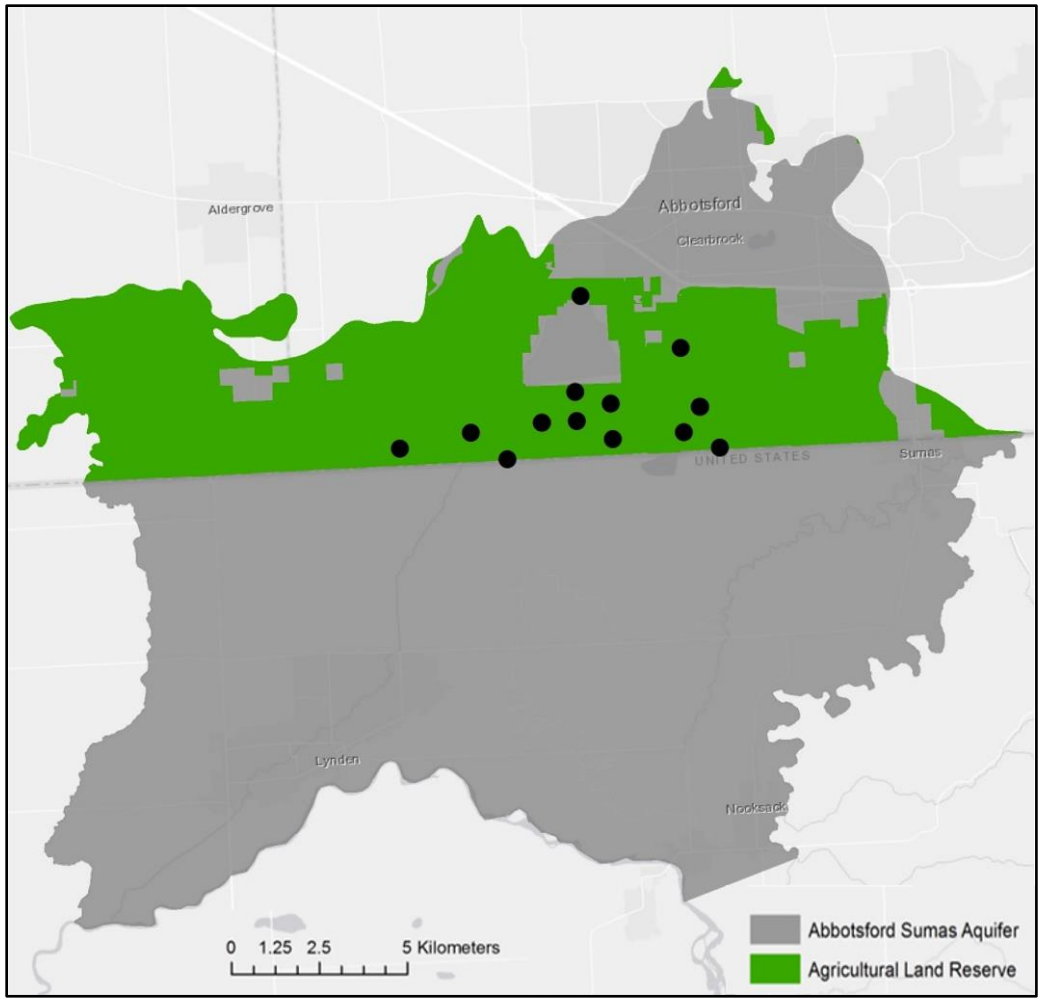

Table A.3. Agricultural Land Reserve (ALR) land that lies within the Abbotsford-Sumas Aquifer (ASA). More than 70% of the Canadian-side of the ASA is comprised of ALR land. Wells in Canada examined in this research are noted by circles.