Phosphorus Legacies and Water Quality Risks: A Vulnerability-Based Framework in Southern Ontario

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

I acknowledge Dr. Philippe Van Cappellen, Dr. Kimberly Van Meter, Dr. Nandita Basu, Dr. Chris Parsons, Dr. Jerker Jarsjö, Dr. Benoît Dessirier, Kevin McKague, and Daniel Saurette who contributed to this research with guidance and as coauthors.

Dr. Kimberly Van Meter provided guidance during the study design, developing methods to interpret and process data. Dr. Philippe Van Cappellen, Dr. Nandita Basu, and Dr. Chris Parsons aided with the writing of the thesis, guiding in decision making for its vision.

Dr. Jerker Jarsjö and Dr. Benoît Dessirier contributed to the development and creation of the Swedish vulnerability model in Chapter 3.

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Abstract

Excess phosphorus (P) loading increases the frequency of harmful algal blooms (HABs), posing severe threats to drinking water security and aquatic ecosystems. Efforts to reduce the inputs of P to Canadian agricultural soils started in the late 1970s-early 1980s, and were initially successful, but surface water P loading became persistent again in the 2000s. HABs were a problem in the southern Laurentian Great Lakes (LGL) before the initial nutrient mitigation efforts, and the re-emergence of HABs in Lake Erie in the 2000s was in part a result of legacy P that had accumulated in soils and groundwater in agricultural watersheds. Legacy P exists as a result of historical inputs of P, typically fertilizer used in excess of crop needs. Consequentially, even after reducing P inputs, legacy P continues to be exported from soils after several decades. In Chapter 2, a large-scale mass balance was conducted for the Ontario watersheds to locate and quantify agricultural and other anthropogenic P inputs from 1961 to 2016, utilizing existing datasets as well as historical reconstructions of P inputs to the landscape. This scale of P mass-balance has not been completed before in Ontario. The mass balance model was implemented into a Geographical Information System (GIS) platform to delineate potential areas with legacy P accumulation and depletion within the landscape. These maps identified areas with high P inputs and large potential stores of legacy P. Historically, southwestern Ontario has had the densest agriculture and populated areas in Ontario and has had high P inputs over time. County-scale trends such as shifts to intensive livestock or crop-based agriculture, or increasing urbanization were also identified. In Chapter 3, the fate and transport of P and the possible development of P legacies was explored in the context of risk. P export is influenced by environmental conditions in soil, as such, there is spatial variance in the likelihood that P will runoff or accumulate in soils. The environmental conditions may therefore be used to represent the vulnerability of the system and the risk to either lose or accumulate P. The cumulative agricultural P surplus map was used in conjunction with vulnerability maps to construct soil P risk maps. Different vulnerability models were explored, and ultimately soil erosion potential maps were used to identify vulnerable areas with a high risk of P losses to surface water and areas with a high risk of P accumulation in soil. It was determined that there was a higher risk of P accumulation in soil along the coast of Lake Erie, and it is possible that P legacies exist in these areas. The results inform nutrient management and abatement strategies and the adaptive implementation of conservation practices.

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Dedication

This thesis is dedicated to Olliver Flynn, my beloved Siberian fluffball cat. You were with me through the entire process, mostly because it is impossible to leave the house without knobs of your fur covering my clothes.

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List of Abbreviations

4Rs	Right source, right rate, right time, and right place, a management practice supported by the Ontario government for the application of fertilizer on agricultural lands
AAFC	Agriculture and Agri-Food Canada
BSE	Bovine Spongiform Encephalopathy, "Mad Cow Disease"
BD	Bulk Density of soil
C	Cropping systems, factor from the Revised Universal Soil Loss Equation
C_a	Animal P consumption, used in the Net Anthropogenic Phosphorus Input formula
C_h	Human P consumption, used in the Net Anthropogenic Phosphorus Input formula
C_i	Total county cropland area for the current crop, used to calculate Crop P production
COFSP	Canada-Ontario Farm Stewardship Program
CP	Conservation Practice
D	Detergent, used in the Net Anthropogenic Phosphorus Input formula
E_a	Animal P excretion, used to calculate net livestock input in the Net Anthropogenic Phosphorus Input formula
EFP	Environmental Farm Plan,
EP	Erosion potential calculated from the Revised Universal Soil Loss Equation
EV	Erosion Vulnerability, variable used to create the intrinsic vulnerability map
FERT	Fertilizer application, used in the Net Anthropogenic Phosphorus Input formula
GIS	Geographic Information Systems
GLWQA	Great Lakes Water Quality Agreement
GRW	Grand River Watershed
HAB	Harmful Algal Bloom
ILO	Intensive Livestock Operation
IPNI	International Plant Nutrition Institute
K	Soil erodibility, factor from the Revised Universal Soil Loss Equation
L	Slope Length, factor from the Revised Universal Soil Loss Equation
LEAP	Legacies of Agricultural Pollutants
LGL	Laurentian Great Lakes
LGLW	Laurentian Great Lakes Watershed
LUA	Land Use Area
M_{AG}	Total estimated surplus mass of phosphorus from agricultural land use
M_{URB}	Total estimated surplus mass of phosphorus from urban land use
MCRA	Multi-Criteria Risk Assessment
MECP	Ontario Ministry of the Environment, Conservation and Parks
N_a	Number of animals for the current livestock type of interest, used to calculate net livestock input in the Net Anthropogenic Phosphorus Input formula

NANI	Net Anthropogenic Nitrogen Input model
NAPI	Net Anthropogenic Phosphorus Input model
NET_a	Net livestock P input, calculated from animal P consumption and production
NFFI	Net Food and Feed Imports
NPS	Non-point Sources
OM	Organic Matter
OMAFRA	Ontario Ministry of Agriculture, Food and Rural Affairs
OMNRF	Ontario Ministry of Natural Resources and Forestry
P	Phosphorus
P*	Support practices, factor from the Revised Universal Soil Loss Equation
P_a	Animal P production, used in the Net Anthropogenic Phosphorus Input formula
P_c	Crop P production, used in the Net Anthropogenic Phosphorus Input formula
PC_i	P content for the current crop of interest, used to calculate Crop P production
P_{DD}	Phosphorus from dishwashing detergent
P_{DL}	Phosphorus from laundry detergent
PI_a	Annual animal P intake for the current livestock type of interest, used to calculate net livestock input in the Net Anthropogenic Phosphorus Input formula
PE_a	Animal P excretion intake for the current livestock type of interest, used to calculate net livestock input in the Net Anthropogenic Phosphorus Input formula
PLUARG	Pollution from Land Use Activities Reference Group
$(Pop)_{County}$	Total population within a county
P_v	Precipitation Variable, variable used to create the intrinsic vulnerability map
R	Rainfall Erosivity, factor from the Revised Universal Soil Loss Equation
P2SW	Proximity to Surface Water, variable used to create the intrinsic vulnerability map
RKLS	A combination of R, K, and LS factors from the Revised Universal Soil Loss Equation
RUSLE	Revised Universal Soil Loss Equation
S	Slope Steepness, factor from the Revised Universal Soil Loss Equation
SD	Slope and Drainage, variable used to create the intrinsic vulnerability map
SLC	Soil Landscapes of Canada, from Agriculture and Agri-Food Canada
SMHI	Swedish Meteorological and Hydrological Institute
STP	Soil Test Phosphorus
TD	Tile Drainage Presence Probability, variable used to create the intrinsic vulnerability map
TRW	Thames River Watershed
VI	Vulnerability Index
WD	Wetland Drainage, variable used to create the intrinsic vulnerability map
Y_i	Annual crop yield for the current crop of interest, used to calculate Crop P production

Chapter 1 General Introduction

Excess inputs of phosphorus (P) to the environment as a result of agricultural intensification and human population growth pose a significant risk to both human wellbeing and the health of aquatic ecosystems. The most visible impact of elevated P levels is the occurrence of algal blooms, including harmful algal blooms (HABs) and the associated expansion of hypoxic "dead zones" in lakes and coastal areas (Carpenter et al., 1998). Other impacts include *Cladophora* nuisance algae blooms (Holeton, 2013), which negatively affect recreation and can reduce lake-side property values. Phosphorus is widely considered to be the primary limiting nutrient in freshwater systems (Smith and Schindler, 2009), and in recent decades, mitigating P inputs from non-point sources (NPS) has been considered by watershed managers to be an essential strategy for controlling freshwater eutrophication (Jarvie et al., 2013b).

The use of fertilizers and manure in agriculture is a major driver of the excess supply of P and other nutrients to water bodies (Butterbach-Bahl et al., 2013; Kirchmann and Thorvaldsson, 2000; Widory et al., 2004). An area of major concern for HABs caused by NPS inputs is the Laurentian Great Lakes (LGL), particularly Lake Erie (Michalak et al., 2013). Both Canadian and United States governments have recognized that high P loadings have led to HABs in the western Lake Erie Basin and have agreed to reduce P loadings by 40% by 2025 (Great Lakes Water Quality Agreement Annex 4, 2012). Conservation practices (CPs) aim to reduce anthropogenic NPS nutrient inputs to surface waters and groundwater, and their implementation has been widespread in the LGL watershed (LGLW) (International Joint Commission, 2014).

However, despite widespread implementation of CPs, adverse effects related to P loading are still seen in the LGL. In addition, today's loadings of P from agricultural NPS reflect not only current farming practices but also nutrient legacies that have accumulated over time in soils, groundwater, sediments, riparian zones, and wetlands as a result of past applications of manure and fertilizer (Jarvie et al., 2013a; Sharpley et al., 2013; Van Meter and Basu, 2015). Legacy nutrients are believed to be a major reason for the observed delays in water quality improvements (Sharpley et al., 2013), among other processes including internal loading from lake sediments (Orihel et al., 2017). The persistence of nutrient release from legacy stores may be offsetting the improvements expected from CPs (Sharpley et al., 2013; Van Meter et al., 2016). The question then becomes – how important are the contributions of these legacies to the export of P from agricultural landscapes, and how long will they persist into the future? To answer this question, the legacy P stores throughout the LGLW and Ontario overall must first be delineated and quantified.

1.1 Ontario Agricultural Land Use History

Ontario agricultural records date back to 1826, meaning that agriculture, as designed by Europeans, has been in Ontario for at least two centuries. Earlier still, First Nations used the fertile soil to grow corn, squash, and kidney beans (Jones, 1946). Agriculture has been a major land use in Ontario throughout its history, and still is today, although the amount of farmland and cropland has decreased since the early 1900s. The total area of crop and pasture in Ontario increased to a maximum in 1891 at nearly 6 million ha, and has since decreased to a relatively stable 4 million ha (Smith, 2015). Total farm area peaked in 1931 at over 9 million ha, 10.1% of Ontario land area, and decreased to just over 5 million ha in 2011, 5.6% of Ontario land area (Smith, 2015). In southern Ontario, the total percent of area taken up by farmland was 60.7% in 1931, and 35.5% as of 2011 (Smith, 2015). The decrease in agricultural land use was driven by increased crop productivity and higher yields, urbanization, and the extension of agricultural development to western Canada (Smith, 2015).

Major agricultural trends in Ontario observed by Smith (2015) since 1920 include a large decrease in the number of cattle raised, and an increase in the amount of corn, wheat, and soybeans grown. These trends not only reflect consumer preferences and economic changes, they also influence nutrient cycling. Further examination of these trends is warranted at a finer scale, especially in southern Ontario. Filling knowledge gaps in agri-environmental land use trends in Ontario has aided in the further development and application of policies and CPs (Smith, 2015). Combining agricultural trend analysis with a P mass balance analysis can help us further understand what drives P cycling in Ontario watersheds.

1.2 Phosphorus Pollution in Ontario

The global trend of P fertilizer use has increased over time, beginning in the 1940s in the post war "green" revolution (Cordell et al., 2009). Combined with the increasing use of P in dishwashing and laundry detergents, the amount of excess P in water systems had become a major issue approaching the 1970s, coined as cultural eutrophication (Hasler, 1969). The consequences of excess P in surface water emerged as nuisance algae, taste and odour problems, and HABs leading to fish die-offs in lakes and coastal areas (Schindler, 1977; Paerl, 1988; Carpenter et al., 1998;

International Joint Commission, 2014). A major example is in Lake Erie, which is the shallowest of the Great Lakes, making it more vulnerable to eutrophication and HABs (International Joint Commission, 2014). To address the problems in the binational waters of the LGL, Canada and the US coordinated their efforts. The Great Lakes Water Quality Agreement (GLWQA) was signed in 1972 and updated in 2012 to improve programs targeted at the integrity of chemical, biological, and physical properties of the lakes (International Joint Commission, 2013). It was known that P was a problem nutrient in freshwater systems when the original GLWQA was signed, however, the significance of P limitation in freshwater and the resulting problems caused were confirmed in the study by Schindler (1977). Emphasis on reducing P inputs from both point sources and non-point sources were included in later iterations of the GLWQA.

1.3 Point Sources

The initial goals of the 1972 GLWQA regarding P involved reducing input from point sources. This included reducing P concentrations in laundry and dishwashing detergents, as well as updating processes in water treatment plants (International Joint Commission, 2014). Policies and regulations were implemented in the United States and Canada to reduce the amount of P from detergents to aid in the growing problem of excess P inputs, which had some success, and was magnified by the updated management policies of other anthropogenic point sources like industrial waste and human waste water (Joosse and Baker, 2011). The control over point sources of P contributed to a successful overall decrease in P loading to the LGL from the 1970s to the 2000s. There were efforts to improve land stewardship over the same period through the adoption of reduced till and no-till practices, however, there was no detectable change in the magnitude of NPS of P during that time (Joosse and Baker, 2011).

Detergent regulations put in place by the Canadian government aiming to reduce P concentrations in detergent began in the 1970s, and the US followed suit (Maki et al., 1984; Litke, 1999). The Canada Water Act of 1970 was passed with the aim of limiting P in laundry detergents and other cleaning agents to 20% P by weight (Booth and Quinn, 1995; McGucken, 1989). This was further reduced to 2.2% by weight in 1972 (McGucken, 1989). The most recent change to the legislation was in 2010, where maximum allowable weight percent of P in both laundry and dish detergents was reduced to 0.5% (Environment Canada, 2011). However, this did not include industrial use of detergent, which was allowed to stay at 2.2% maximum allowable weight percent of P. The change in regulation was set to be in effect by July 1, 2010 and was announced with enough time for manufacturing companies to make changes to their detergent formulas, with many removing P completely, opting to market detergents with “zero phosphates”.

1.4 Non-Point Sources

Following the success in point source reduction in the LGLW, NPS became the main nutrient input to the freshwater system (Bennett et al., 1999), and are now the focus of CPs to achieve water quality goals. A major source of NPS nutrient inputs in southern Ontario is agricultural activities (Bennett et al., 1999). In 1976, it was estimated that 24% of P entering the LGLW was from agricultural sources (van Bochove et al., 2011). In Lake Erie alone, this proportion increased to 60% for rural NPS of P (van Bochove et al., 2011). NPS are more difficult to control, as their behaviour is more difficult to predict and measure on land management scales. Loading from NPS is also influenced by spatially variable fertilizer inputs, land use changes, and weather conditions such as rainfall frequency. In addition, as rainfall frequency and intensity varies with climate change, the uncertainty of P loading increases (Carpenter et al., 2017). Monitoring NPS P loading requires watershed-scale studies (Carpenter et al., 1998), increasing the difficulty of understanding the effectiveness of mitigation strategies. The variety of environmental influences involved makes it more difficult to detect improvements in P loading on a large scale.

In the agricultural dominated region of southern Ontario, fertilizer application was greater than crop uptake for decades (International Plant Nutrition Institute, 2013a). However, because of awareness and mitigation efforts, phosphorus fertilizer application was successfully decreased by 30.1% between 1981 and 2011 (Smith, 2015). Details regarding the initiation of these and many other efforts can be found in the reports from the Pollution from Land Use Activities Reference Group (PLUARG), created by the International Joint Commission to address Article VI of the GLWQA of 1972 (PLUARG, 1974). The widespread implementation of CPs and educational programs was successful in reducing total agricultural P inputs to the soil, but there were still problems being observed in the LGL into the 2000s (International Joint Commission, 2014). It is probable that the historical long-term excess use of fertilizer has led to P accumulation in soils and groundwater, which has the potential to be mobilized long after excess inputs have stopped (Powers et al., 2016).

1.5 Fate and Transport of P

Accumulated P represents the past activity and anthropogenic treatment of the land, which has now been inherited by landowners, government, and other stakeholders as a problem that needs fixing. Phosphorus that has accumulated from historical behaviour is now referred to as legacy P, as it represents the legacy of nutrient management of the past (Sharpley et al., 2013). Legacy P occurs when P attaches to sediments for long periods of time, accumulating in soil, riparian zones, and wetlands, or alternatively being stored in slow moving aquifers.

Legacy P stored in sediments is continuously mobilised over time and exported to surface waters through a number of processes (Figure 1-1). For example: particulate P can become a source of dissolved, bioavailable P through mineralization; decreasing inputs can lead to groundwater and surface water having lower background P concentrations, changing the diffusion gradient between water and sediment, and causing previously immobilized P to be released (Sharpley et al., 2013); P attached to sediments through adsorption can be released if conditions become anoxic, or new ions are introduced in the sediment pore water (Figure 1-1).

Phosphorus can move to surface water through overland flow during precipitation events or through subsurface flow (Figure 1-1). Transport mechanisms include suspension in water as dissolved P, particulate P, or as P attached to sediments. Although P can move in the subsurface in groundwater, or through tile drains underneath agricultural land, P tends to attach to sediments underground and is slow to completely move out of the subsurface.

Furthermore, soil erosion transports P to surface water in the form of immobilized P attached to eroding sediments (Figure 1-1), and this process increases the vulnerability of P losses in that area. Alternatively, a lack of soil erosion may lead to P accumulation in the subsurface. Over time, areas of legacy P are sinks of P which may eventually become sources of P, as the soil is exposed to erosion or other mechanisms which lead to the transport of P, such as when as redox conditions change, or when P attachment sites saturate (Figure 1-1).

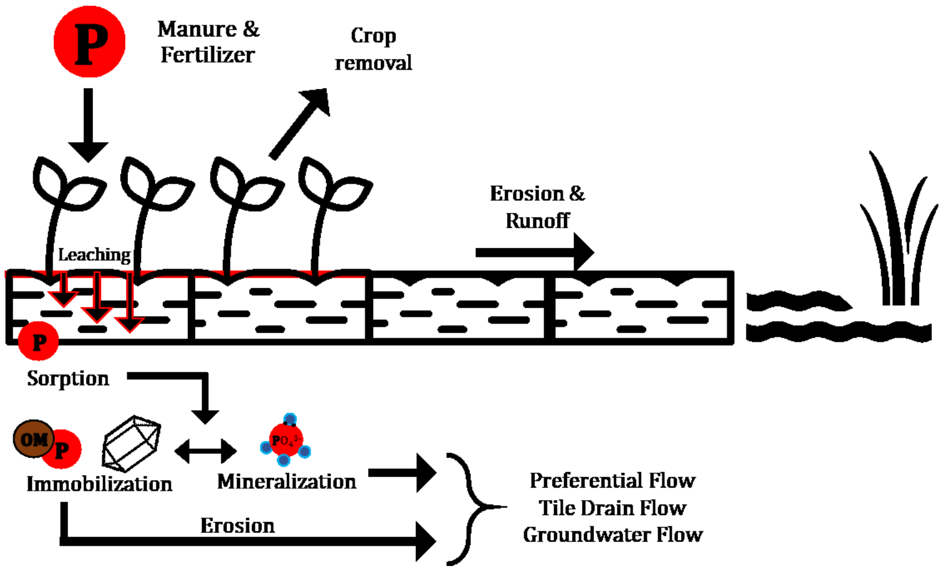


Figure 1-1. Phosphorus fate and transport mechanisms in agricultural soils. P represents phosphorus and OM represents organic matter. Adapted from Shigaki et al. (2006) and Sims and Kleinman (2005). Infographic modified from images from the Noun Project by: Davis; Irene; Lagunov; MFRA.

Continuous output of P from P saturated soils, groundwater, and other sources, can persist for decades, offsetting the implementation of CPs. The result is an illusion of failure of the CPs, and that alternative methods would have to be developed. However, the behaviour of legacy P can explain this offset. Addressing legacy sources of nutrients involves more CPs that specifically target storage areas and transport pathways. For example, building buffer zones and wetlands can aid in creating temporary storage sites for excess P, and prevent it from entering surface water systems. However, it is important to note that some CPs can have unintended consequences such as converting particulate P to dissolved P (Sharpley et al., 2013) and should therefore only be applied in appropriate areas.

1.6 Conservation Practices

Efforts to reduce NPS P inputs in agriculture in the LGLW have been ongoing since they were identified as a problem in the 1970s. Conservation practices were introduced and adopted by individuals all over Ontario and have since become a significant part of mitigation efforts. Some commonly used CPs in the LGLW include reduced P in feed, regulated manure and fertilizer application, improved manure management, riparian zone implementation, over-winter cover crops, on-farm wetland construction, and conservation tillage (Bruulsema et al., 2011; International Joint Commission, 2014). In addition to these CPs, the Nutrient Management Act was introduced in 2002 (Government of Ontario, 2018), which targeted agricultural inputs of nutrients with a focus on livestock (van Bochove et al., 2011).

For nutrient management, one strong approach for reducing loads is to retain surplus nutrients in their current location. CPs that achieve this are referred to as transport CPs (Sharpley et al., 2006) and one example is on-farm constructed wetlands. Down gradient storage systems keep P on the farm premises which can then be used and stored by the wetland vegetation (Sharpley et al., 2006). Small wetlands are critical in retaining P (Cheng and Basu, 2017) and constructing on-farm wetlands has the potential to increase the P storage capacity on the landscape.

Cover crops are another transport CP that can be used in agricultural areas to remove excess P over-winter, prevent P losses via surface runoff, and reduce erosion in heavily erodible areas (Dodd and Sharpley, 2016; Sharpley et al., 2006). However, cover crops may increase the presence of dissolved P, therefore, their implementation must target areas at higher risk of erosion and avoid areas where soil may have high concentrations of P.

Other transport CPs include riparian zones, conservation tillage, and buffer strips, all of which are designed to prevent P movement to at-risk water bodies (Dodd and Sharpley, 2016; Sharpley et al., 2006). The most effective use of these CPs is critical in lowering costs and improving efficiency to achieve the 40% reduction loading goals set by Canada and the United States.

Plans to increase environmental stewardship within the agricultural community initiated by farmers and were officially started with Ontario's Environmental Farm Plan (EFP) Program. Beginning in 1992, farmers were introduced to this non-regulatory framework that would allow them to assess their personal environmental stewardship on their farm, and practices that they could implement within their own EFP to mitigate potential issues. The program is voluntary, confidential, and emphasizes education and awareness surrounding environmental concerns (Plummer et al., 2007). Increased financial support began in 2005 when the EFP program was connected with the Canada-Ontario Farm Stewardship Program (COFSP) (Ontario Federation of Agriculture, 2011). Having gone through three iterations (the second in 1996, and the third in 2004), this EFP program currently provides educational and financial support up to $30,000 as part of the COFSP (Ontario Federation of Agriculture, 2011). The financial aid supports the implementation of CPs by incentivizing farmers to adopt the practices and enables individual environmental stewardship.

A major initiative in the LGLW to reduce P inputs to Lake Erie is the 4Rs nutrient management program (right source, right rate, right time, and right place) developed by the International Plant Nutrition Institute (IPNI) (International Joint Commission, 2018). Changing the method of P application (source) can affect the retention of P in the soil, for example: using banding application rather than broadcast application can reduce P stratification in soil and increase P retention (Grant et al., 2019; IPNI, 2012). Applying fertilizers at the right rate by conducting soil tests and monitoring soil P levels can help with both reducing excess P and mitigating P deficiencies in agricultural soils. Right rate highlights the importance of soil testing to optimize nutrient use efficiency. The potential for P runoff from precipitation events and the timing of crop needs are both important factors to consider for the right timing of fertilizer applications. The ideal, although currently impractical, timing of fertilizer applications would be to apply at planting (IPNI, 2012). Instead, optimizing application timing by avoiding high runoff events in late fall, early spring, and winter, as well as applying as close as possible to the time of planting, will aid in the reduction of P losses to surface water (IPNI, 2012). Finally, right placement of fertilizer, e.g. below the top five centimeters of soil, can reduce P losses in runoff (IPNI, 2012). In addition, placing P fertilizer below the soil surface brings the nutrients closer to plant roots. Right placement can also include avoiding sensitive areas, or areas with high runoff potential, close to surface water or drainage areas (IPNI, 2012). The present study will aid in identifying the right application rate of fertilizers by identifying areas with high long-term P inputs, at a quaternary watershed scale, highlighting areas where soil P testing is critical before fertilizer application.

There is a high prevalence of CPs throughout southwestern Ontario and they have experienced increased implementation, especially in areas with concentrated agricultural activity (Woyzbun, 2012). Applying CPs can be expensive and time consuming, therefore it is necessary to use them efficiently. It is critical to identify optimal locations to maximize CP effectiveness.

1.7 Objectives

The overall objective of this thesis was to locate regions in Ontario with high P input over time to aid in identifying possible locations with legacy P buildup. Working with long-term historical data, the following questions were asked: How has legacy P accumulation differed spatially in Ontario? And how can understanding and quantifying the temporal evolution of P legacies help inform the targeting and implementation of CPs? Within the thesis, these questions will be addressed with the following specific objectives:

1) Conduct a long-term anthropogenic P mass balance in Ontario counties and produce a series of maps that spatially represent the annual and cumulative mass balance.
2) Examine the regional trends of P sources considered in the mass balance to understand the drivers of P legacy risks in the different agricultural regions of Ontario.
3) Use the mass balance findings to produce risk maps that delineate the agricultural areas susceptible to the erosion of P in soil or accumulation of P in soil to aid in identifying areas that would be at risk of accumulating P legacies, and areas that would be at risk of eroding previously accumulated P legacies.

The project outcomes included a Geographical Information System (GIS) informational time series map and two risk maps that will be made publicly available and could be used by partner organizations. This work contributed to the Lake Futures and Legacies of Agricultural Pollutants (LEAP) project (https://uwaterloo.ca/legacies-of-agricultural-pollutant/).

Chapter 2 Annual and Cumulative Anthropogenic Phosphorus Mass Balance in Ontario

2.1 Introduction

Unbalanced P cycling has the potential to significantly negatively impact the environmental health of surface water ecosystems. Several mitigation efforts have been implemented in Ontario to more effectively manage nutrient movement in agricultural regions. However, as previously discussed, legacy P has the potential to offset these efforts (Sharpley et al., 2013; Van Meter et al., 2016). Therefore, developing a better understanding of the spatial distribution of P legacy development could aid in the more accurate targeting and mitigation of excess P. An initial step in identifying possible legacy P accumulation locations is identifying areas that have a historical surplus of P by conducting a long-term mass balance analysis.

The approach used to conduct the mass balance was the Net Anthropogenic Phosphorus Input (NAPI) model (Russell et al., 2008; Zhang, 2016), which was applied at a county scale for all of Ontario. The mass balance model was integrated into a GIS platform to develop a user-friendly map time series that delineated the spatial and temporal evolution of P inputs and possible legacies in Ontario counties and watersheds. An estimated NAPI value greater than zero represented a surplus of P in that area, and a larger surplus would effectively increase the risk of legacy P development.

Mass balances must consider the inputs and outputs within a system. Notably, there are several anthropogenic P input and output mechanisms in the agro-ecosystems of Ontario. One significant output, or export, in agricultural land use areas is crop production, which removes P from agricultural fields. There is less cropland today than there was 60 years ago, but the amount of crop mass grown has increased (Smith, 2015). Crop yields have increased over time, leading to increased production, and higher efficiency in farming (Smith, 2015). Increased crop yields combined with decreased fertilizer application may have led to increased crop uptake of historical excess P within the LGL. However, it may take time before the benefits are detectable, as there are time lags associated with historical P accumulation, potentially delaying the response time by several decades (Sharpley et al., 2013).

Phosphorus in both dissolved and particulate forms can be removed from soil and transported to rivers and lakes through overland flow, groundwater flow, and erosional processes. However, historical river export data are sparse in Ontario, and total P export by rivers would be difficult to estimate on a large scale. Consequently, river export of P was not estimated, instead, crop uptake was used to represent the major output of P from the system. Additionally, river export data would likely represent a minor portion of the mass balance: a study in the St. Lawrence Basin found that river export represented only 5% of the NAPI balance (Goyette et al., 2016). A smaller scale study found that river exports increased over time and could represent up to 10% of the mass balance (Pardy, 2019). Despite this, the majority of P inputs have not yet been transported to surface waters and these river outputs have the potential to continue for many years after P inputs have been reduced as result of excess P accumulation in the subsurface.

In this chapter, analysis of the data led to the identification of potential legacy accumulation zones, interchangeably referred to as areas with a P surplus, meaning that more P was applied to the land than was used by crops. The scale and resolution of the approach have not been done before in a legacy context for Ontario as a whole, but smaller scale projects have been conducted (van Bochove et al., 2011; MacDonald and Bennett, 2009; Zhang, 2016). Other similar studies such as Joosse and Baker (2011) have examined the temporal changes in P concentrations in the Great Lakes and connected tributaries, however, the present study differs by including a spatial component. In addition, a regional analysis was conducted to further our understanding of the drivers of P cycling in different agricultural regions in Ontario.

2.2 Methods

The primary objective was to map the potential distribution of agricultural P legacy development within Ontario through a large-scale spatio-temporal mass balance approach (Van Meter et al., 2017). A long-term (1961-2016) P mass balance for the study area was carried out based on existing datasets as well as historical reconstructions of P inputs to the landscape. Reconstructed P accumulation trajectories were visualized and analyzed in ArcMap, an ArcGIS application tool.

The NAPI model was applied to the entire province, and a cumulative map was produced in addition to the map time series to highlight areas of high anthropogenic P input. The spatial information was useful in identifying areas that could be major legacy sources of P and could inform government and decision makers about the P affecting the LGL, which has become an enormous governmental concern (Michalak et al., 2013; International Joint Commission, 2014; Powers et al., 2016). It is important to note that the map indicated historic P input location, not current P location, but the data could be used in mechanistic models as input parameters or initial conditions. The information available in the map time series and the cumulative input map enhance knowledge about legacy P locations in Ontario and can, therefore, inform CP implementation strategies.

2.2.1 Administrative Units

Ontario's administrative divisions are complex in their classification and purposes but can be grouped into general categories. Below the province and territory level of division, Statistics Canada refers to the next tier as census divisions, comparable to counties, which can be further divided into census subdivisions, comparable to townships. The agricultural census and population census data were available on the subdivision level, but in the present study, P inputs were determined at the division scale. Additionally, the number and size of divisions, hereafter referred to as counties, have changed over the course of Ontario's history, which was accounted for in this study by standardizing counties to their location in 2011, the same year as the most recent available county shapefile (Figure 2-1). The assumption was that the borders of counties in 2011 were the same in previous years; data from subdivisions that were moved or no longer exist were included in the county that subdivision's location resided in 2011.

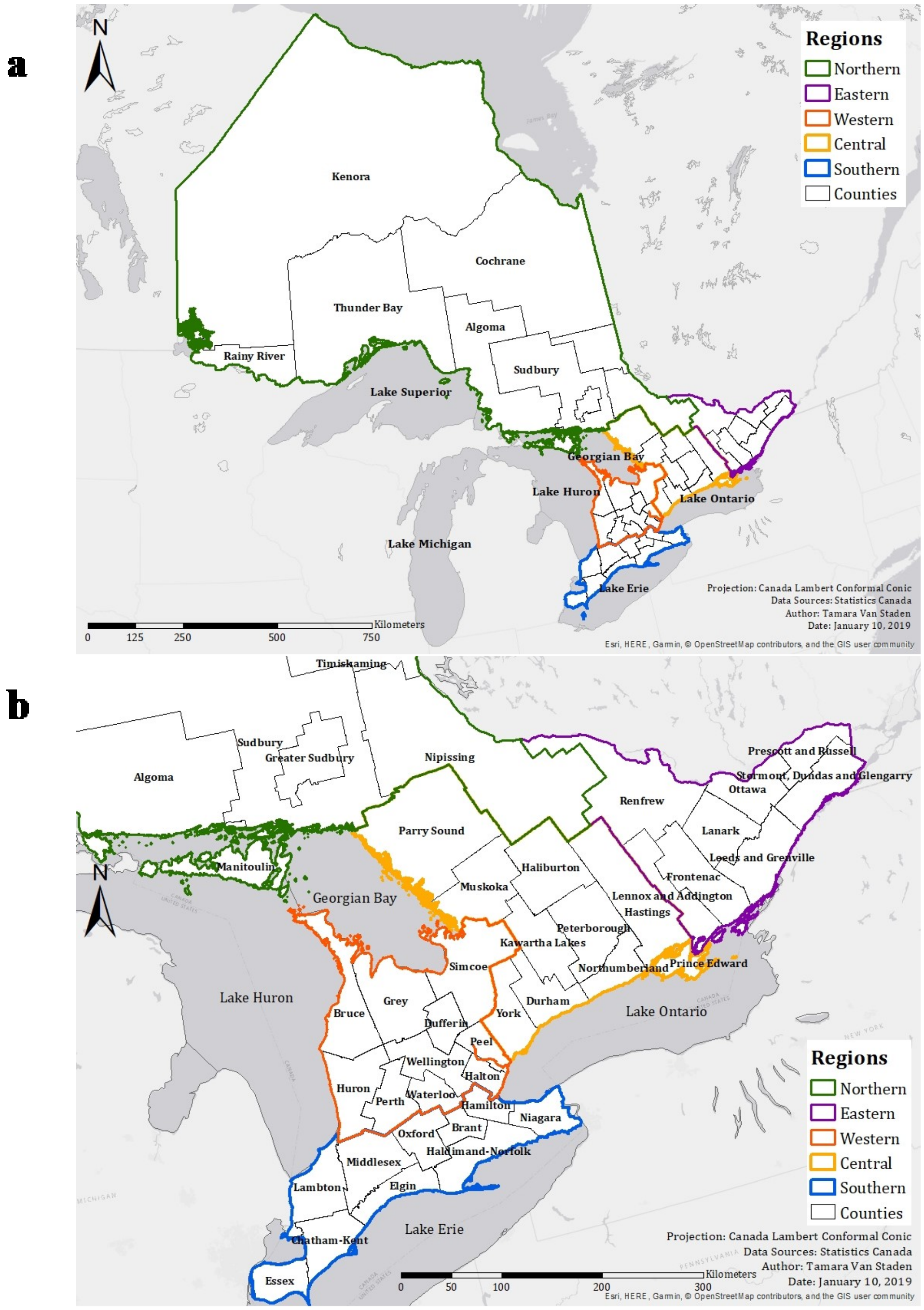


Figure 2-1. Distribution of counties within agricultural regions in a) Ontario and b) southern Ontario in 2011.

2.2.2 Watershed Description

Most of the Canadian watersheds that drain into the LGL are located inside Ontario and are divided among four of the five lakes. No part of the Lake Michigan watershed is located within Canada and was therefore not included in this study. The total watershed area of Lake Superior, Lake Huron, Lake Erie, and Lake Ontario is 397,038 square kilometers, about half of which is located in Canada (Government of Canada, 2015). Lake Superior and Lake Huron respectively have 64.9% and 68.6% of their drainage areas located in Canada (Table 2-1). Lake Erie and Lake Ontario have watershed area located in the United States, with only 28.4% and 44.1%, respectively, of their drainage area within Canada. The total and Canadian drainage areas of the lakes of interest can be found in Table 2-1. On the quaternary scale, there are over 1000 watersheds in the LGLW (Figure 2-2). The largest quaternary watershed is in the Lake Superior basin, and is 605 km^2, and the smallest is in the same basin and is 0.4 km^2 (OMNRF, 2010).

Table 2-1. Total drainage area of the Great Lakes shared by Canada and the United States. Modified from Robertson and Saad (2011).

	Superior	**Huron**	**Erie**	**Ontario**	**Total**
Total Drainage Area (km^2)	124,155	131,614	77,519	63,750	397,038
Canadian Drainage Area (km^2)	80,561	90,245	22,031	28,089	220,926
Canadian Percent of Total (%)	64.9	68.6	28.4	44.1	55.6

The sedimentary and topographic characteristics of Ontario were shaped mostly by ancient glacial activity. The majority of southern parts of Ontario are covered in glacial moraines made of sandy and silty till, while a significant portion of northern Ontario has exposed bedrock, lacustrine deposits, and wetland organic deposits (Baldwin et al., 2000). The Soil Landscapes of Canada characterize southern Ontario soils as Gleysolic and Luvisolic, both of which support widespread agricultural activity, although poorly drained (Baldwin et al., 2000). The soils of northern Ontario vary significantly and range from excessively well drained in the Sudbury region to very poorly drained in the northern wetlands (Baldwin et al., 2000).

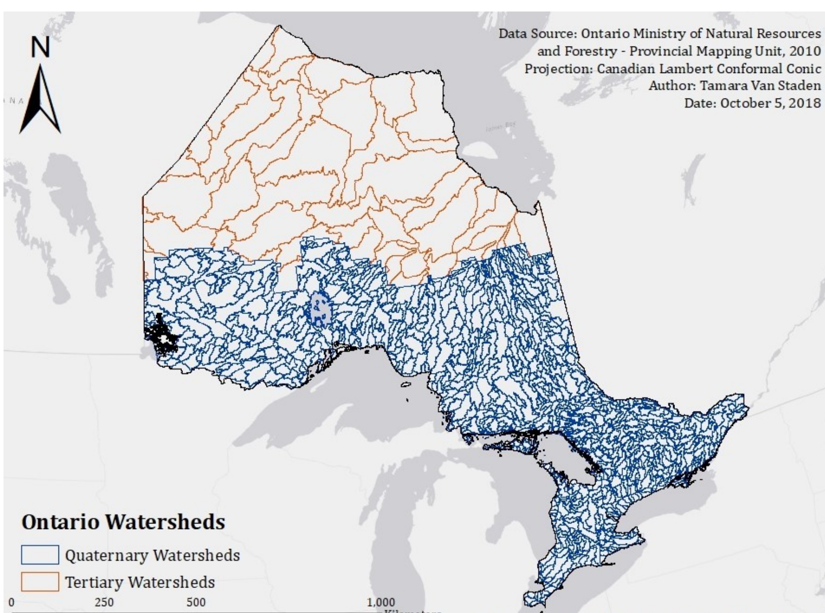


Figure 2-2. Map of the quaternary and tertiary watersheds in Ontario. Data from OMNRF (2010).

2.2.3 Land Use

Ontario land use in the LGLW includes agriculture, forested land, urban areas, and wetland areas. Northern Ontario is mostly made up of forested land and wetland area, and, as previously stated, southern Ontario is dominated by agriculture (Robertson and Saad, 2011). One third of the land in the LGLW is used for agricultural purposes, which makes up most of Ontario's agricultural industry (OMAFRA, 2016). Major urban areas include: Toronto and the Greater Toronto Area, Hamilton, Kitchener-Waterloo, Guelph, London, and Sudbury (Robertson and Saad, 2011).

2.2.4 Net Anthropogenic Phosphorus Input (NAPI) Methodology

The NAPI methodology was used to assess phosphorus inputs in Ontario watersheds, and subsequently, the LGLW. The NAPI model was built off of the NANI (nitrogen) model, originally introduced by Howarth et al. (1996), and adjusted to consider processes that affect the anthropogenic P cycle. There are three main components in the NAPI model: individual detergent use (D, dishwashing and laundry), fertilizer use (FERT), and net food and feed P imports (NFFI). The following description of the NAPI methodology was built directly on the methods used in the Master's thesis written by X. Zhang at the University of Waterloo (Zhang, 2016). The sum of the components listed above represents the net anthropogenic P input in this model:

$$NAPI = FERT + NFFI + D \qquad \text{Equation 1}$$

2.2.4.1 Detergent Use

Detergents used for laundry and dishwashing contain P, and have been noted as a large source of P in urbanized areas (Han et al., 2011). They have been included in the NAPI model in order to quantify all anthropogenic sources of P, even though detergent inputs do not necessarily directly interact with the landscape. The amount of P in detergents has changed over time as a response to water quality and eutrophication problems (Litke, 1999). Laundry and dish detergent per capita usage were determined by assuming that each detergent type was used on a widespread scale beginning in 1935, after their invention (Litke, 1999; Han and Bosch, 2012). The amount of detergent used per wash event increased for laundry detergent to a stable value per capita as of the late 70s and decreased for dish detergent to a stable value per capita in the 90s. This information, combined with the 2010 regulations, was applied to the methodology used by Han and Bosch (2012) to estimate the kg P/capita/year for both types of detergent (Figure 2-3, see values in Table A-2 in Appendix A). Note that the values changed over time due to regulations and increased proportion of households with dishwashers. It should also be noted that several companies now have “zero phosphate” detergents. County scale detergent P use was determined using the annual per capita detergent use value and the counties’ population in the formula from Zhang (2016):

$$D = (P_{DL} + P_{DD}) \cdot (Pop)_{County} \qquad \text{Equation 2}$$

where P_{DL} is the annual per capita P from laundry detergent, P_{DD} is the annual per capita P from dishwashing detergent, and $(Pop)_{County}$ is the population of the county in the same year.

Although P from detergents will not end up on land in the same way as fertilizer and manure, it was included in the NAPI analysis as a spatial indicator of P sources, both point and non-point, into the watershed systems. It is recommended that land-based analyses following this study should ignore point source inputs such as detergent as they enter the system as part of urban wastewater and will be either removed by wastewater treatment processes and sent to landfill or pumped directly into surface water.

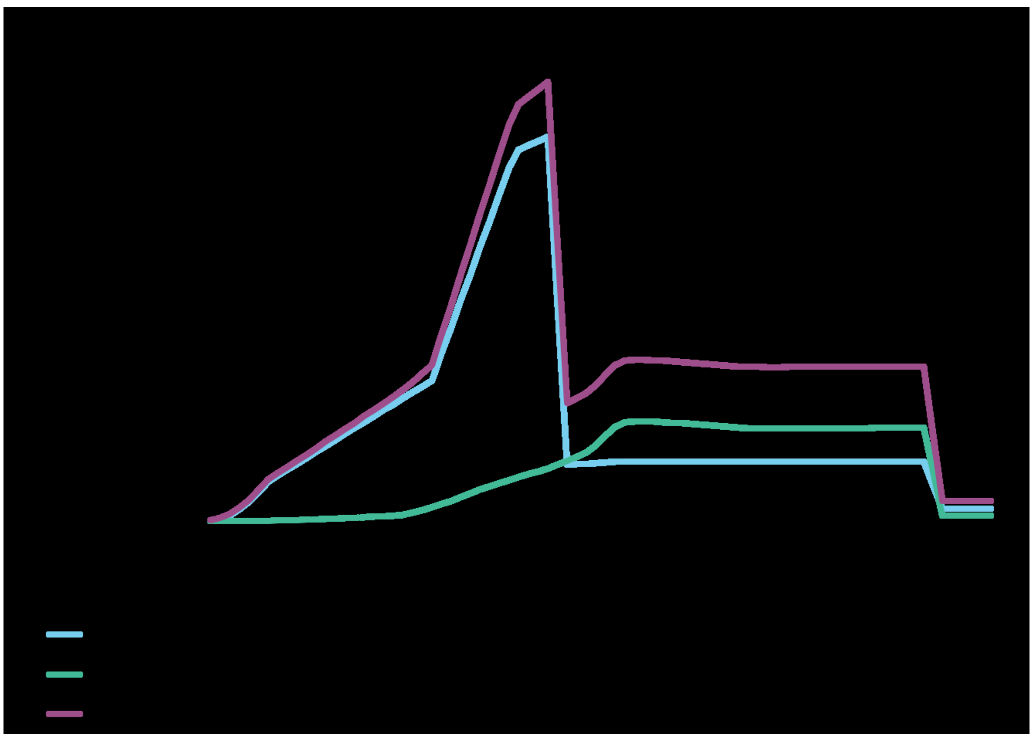


Figure 2-3. Total mass of P sourced from detergents based on per capita domestic detergent use. Methodology and data from Environment Canada (2011) and Han and Bosch (2012).

2.2.4.2 *Fertilizer Use*

Fertilizer sales were not provided on a county scale, instead, province-wide fertilizer sales were used to estimate county-scale fertilizer use (Figure 2-4). The amount of fertilizer sold to each county was estimated based on the county’s total fertilized area provided in the agricultural census and the total provincial fertilizer sales. It was assumed that fertilizer was applied evenly on all fertilized area, although it should be noted that different crops have different fertilization requirements, and crop rotations and productivities vary spatially. The formula to calculate the county scale fertilizer applied in each year is as follows, modified from Zhang (2016):

$$FERT = F_{sales}(tonne\ P_2O_5) \times \frac{FA_{county}}{FA_{province}} \times 0.906 \times 436.4 \frac{kg\ P}{tonne\ P_2O_5} \qquad \text{Equation 3}$$

where F_{sales} is the total fertilizer sales in Ontario for the year of interest, and FA represents the total fertilized area for the county of interest and the Province of Ontario. The final units of FERT were kg P. Fertilizer sales were reported as tonnes of P_2O_5, which is equal to 436.4 kg P per tonne P_2O_5, and the proportion of commercial fertilizer used was estimated to be 90.6% (Zhang, 2016).

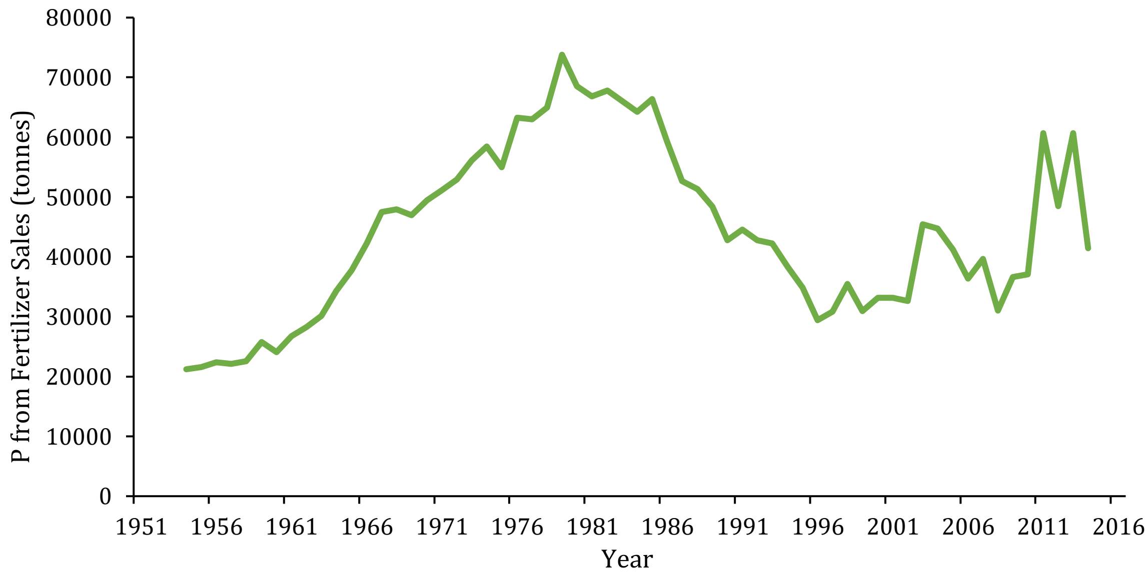


Figure 2-4. Ontario phosphorus fertilizer sales. 1954 – 2007 data from the International Plant Nutrition Institute (IPNI) (2013) and 2008 – 2016 data from the Canadian Fertilizer Institute (2015).

The 1981 census provided mass of applied fertilizer data on a county scale, which was used to test the accuracy of Equation 3. When census values were compared to estimated values, the difference between the estimated percent mass distribution and the actual mass distribution was within 1% for most counties, suggesting that the estimation method was reasonable. Any difference was a result of an under-estimation of fertilizer use in the southern region (Table 2-2), one of the most agriculturally dominated regions. According to an Agriculture and Agri-Food Canada (AAFC) survey in 2010, southern Ontario had approximately the same amount of agricultural land use area as western Ontario, double the amount as central and eastern Ontario, and nearly four times as much as northern Ontario. This method therefore produced a conservative estimate of fertilizer use in southern Ontario, and fertilizer inputs have the potential of being significantly higher there.

Table 2-2. The difference between the estimated and actual fertilizer mass percent distribution in each agricultural region in 1981. The regions' fertilizer mass was divided by the total fertilizer mass in Ontario.

Region	Estimated (%)	Actual (%)	Difference
South	42.16	53.88	-11.72
West	33.28	28.15	5.13
Central	10.72	9.32	1.40
East	10.36	7.59	2.77
North	3.48	1.06	2.42

2.2.4.3 Net Food and Feed Imports (NFFI)

There are four components in NFFI: consumption of P by humans (C_h), consumption of P by animals (C_a), production of P by animals (P_a), and production of P by crops (P_c) (Zhang, 2016). Whether NFFI was positive or negative represented whether the system needed to import the P required by human and animal consumers (positive), or whether the system was producing enough P for the consumers and had excess to export (negative). It was assumed that there was a 10% loss of P for both livestock and crop products as a result of processing, waste, and pest consumption based on previous studies on product use efficiencies and pest-related losses in storage (Jordan and Weller, 1996; Zhang, 2016). All four were combined to calculate the net input of P from the production side of the agricultural industry in the following formula:

$$NFFI = C_a - P_a + C_h - P_c \qquad \text{Equation 4}$$

2.2.4.4 Human Consumption

Human consumption (C_h) was based on the available P in Canadian food per person per day. These data were made available by Statistics Canada as mg P available in food from 1981 to 2009, adjusted for losses (Statistics Canada, 2009). Available P was extrapolated and adjusted to kg per capita per year (Table 2-3). Each annual value was multiplied by the corresponding year's population for each county to get the total human consumption in kg for that year.

Table 2-3. Human available phosphorus in food (Statistics Canada, 2009). Values before 1981 and after 2009 were linearly extrapolated.

Available P	1961	1966	1971	1976	1981	1986	1991	1996	2001	2006	2011	2016
Day (mg/capita/day)	1130	1135	1140	1145	1150	1154	1123	1170	1227	1191	1156	1118
Year (kg/capita/year)	0.413	0.414	0.416	0.418	0.420	0.421	0.410	0.427	0.448	0.435	0.422	0.408

2.2.4.5 Crop Production

Crop production (P_c) of P was estimated using the provincial average of the annual yield for each crop and multiplying that by the average P content for that crop (Equation 5). The total yield was calculated by multiplying the cropland area for major crops from the agricultural census by the average annual yield per hectare (Figure 2-5). It was assumed that yields were constant throughout the province as finer-scale data were unavailable for most census years (see yield values in Table A-1 in Appendix A), and that crop P content stayed constant over time. It is possible that crop P content has changed over time as a result of varying crop varieties and improvements in crop yields, but these data either did not exist or were not readily available.

Major crops in Ontario include corn, wheat, and soybeans, among others. Corn and soybeans are categorized as row crops, and they experienced a significant yield increases after 2001 (Figure 2-5). The steep increase in row crop yields correlated with an increase in wheat yields (Figure 2-5); around this time, wheat was introduced as a recommended crop in crop rotation to improve soil health (Brown et al. 2017; Smith, 2015). It is possible that the inclusion of wheat in the recommended crop rotation contributed to the significant increase in row crop yields. Horticultural crops were not included in analysis because the total crop area of field crops made up more than 95% of the total cropland in Ontario. Two counties that could be affected by this are Niagara county and Prince Edward county because of their wine industries. The total crop yield in kg for each crop in each county was multiplied by the average P content for that crop (Table 2-4), and all crop values were summed to estimate the total crop production of P for that county.

$$P_c = \sum_{i=crop}(CA_i \times Y_i \times PC_i) \times 0.9 \qquad \text{Equation 5}$$

where CA is the total area of the current crop in the county, Y is the yield of that crop for the current year, and PC is the P content of the crop. The value is multiplied by 0.9 to estimate the mass after a 10% loss.

Table 2-4. P content of major crops in Ontario. The value for canola is from Hong et al. (2012), all other values from Zhang (2016).

Crops	g P / kg Yield
Wheat (includes winter, spring, buckwheat, and barley)	3.67
Rye	3.15
Oats	3.24
Mixed grain	3.22
Corn for grain and fodder	2.76
Canola	5.65
Soybeans for beans	5.88
Flaxseed	3.22
Dry white beans and other coloured beans	5.88
All hay	4.54
Potatoes	0.59
Tobacco	4.41

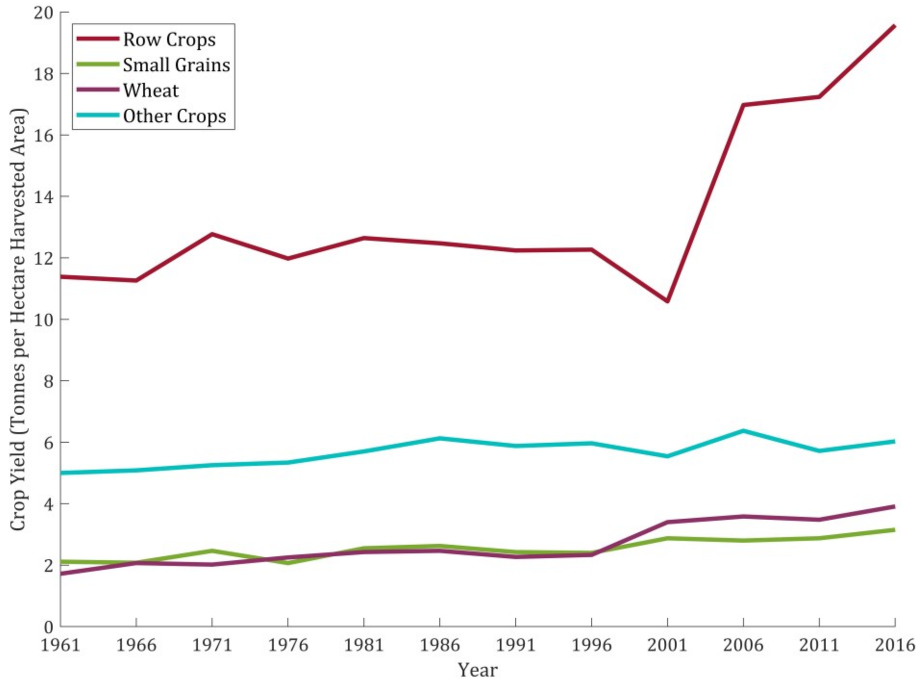


Figure 2-5. Annual Ontario crop yields for the crops examined in tonnes per hectare. Row crops include corn and soybeans; small grains include rye, oats, barley, and mixed grains; and other crops include canola, flaxseed, beans, hay, potatoes, and tobacco. Data from OMAFRA (2018).

2.2.4.6 Animal Consumption and Production

Animal P consumption (C_a) and production (P_a) were estimated using county-scale agricultural census livestock data. Recoverable manure was not estimated in this analysis, instead, a pseudo representation of manure estimated as a net excretion of P was used that accounted for losses after the processing of animals (Zhang, 2016). First, animal P consumption (C_a) and excretion (E_a) were estimated: the number of each livestock type was multiplied by the annual P intake and excretion values from Hofmann and Beaulieu (2006) (Equation 6 and Equation 7). It was assumed that livestock P intake and excretion stayed constant over time due to data limitations.

$$C_a = N_a \times PI_a \qquad \text{Equation 6}$$

$$E_a = N_a \times PE_a \qquad \text{Equation 7}$$

where N_a is the number of number of animals, and PI_a and PE_a are respectively the P intake and P excretion values of that livestock type (Table 2-5).

Second, livestock P production (P_a) was calculated by subtracting animal P excretion (E_a) from animal P consumption (C_a). Animal P production was defined as the P that was removed from the system by retention in the animal. The amount of P that remained in the animal was considered an output from the system. The value was multiplied by 0.9 to estimate the mass after a 10% loss.

This did not apply to humans because it was assumed that they excrete 100% of P consumed (Han et al., 2013).

$$P_a = (C_a - E_a) \times 0.9 \qquad \text{Equation 8}$$

The net animal P inputs for each county were estimated by subtracting the calculated production (P_a) from the calculated consumption (C_a) for each livestock type, simplified in the following formula:

$$NET_a = \sum_{a=animal}(N_a \times (0.1PI_a + 0.9PE_a)) \qquad \text{Equation 9}$$

Table 2-5. Annual intake and excretion values for livestock in Ontario. Values for horses are from Han et al. (2013), all other values from Hofmann and Beaulieu (2006).

Animal	Intake kg P / animal / year	Excretion kg P / animal / year
Beef cows	27.68	21.3
Milk cows	33.16	26.8
Total heifers	20.45	14.1
Steers	20.08	15.2
Bulls	38.66	24.4
Calves	8.06	6.9
Hogs and pigs	6.61	3.1
Horses	13.15	11.7
Sheep and lambs	2.12	1.4
Hens and chickens	0.21	0.123
Turkeys	0.72	0.57

2.2.4.7 Milkhouse Waste

Milkhouses are another source of phosphorus loading in agricultural watersheds. This source was not included in NAPI in previous analyses. Long term practices at dairy operations include washing the floors of the milkhouse for sanitary purposes. The result of this cleaning is an additional input of P from the milk waste, as well as the detergent used to clean the floors. Detergent is the more significant contributor to milkhouse waste P with 93% of the total P coming from the detergent and acid rinse cycle (Allaway, 2003). This increased over time as the practice of cleaning of milk houses with detergent became more widely adopted until it was completely adopted after the Milk Act was implemented in 1990 (Government of Ontario, 1990). In southwestern Ontario, the mass of P per dairy cow added as a result of milkhouse waste ranged from 0.36 kg P/cow/year to 1.14 kg P/cow/year (Allaway, 2003). An average value of 0.77 kg P/cow/year was estimated using the values in Allaway (2003) and was used as the maximum amount per head as of 1991. The average P input per dairy cow was estimated for each census year based on a linear increase starting in 1949 until 1991, after which, it was assumed that P input per cow was constant at the maximum value of 0.77 kg P/cow/year (Table 2-6). In 1949, it was assumed that there was 0 kg P added per cow, as detergent cleaning practices were likely first introduced in regulation at this time (Clegg, 1956).

Table 2-6. Mass of phosphorus per dairy cow due to milkhouse waste in census years (Allaway, 2003).

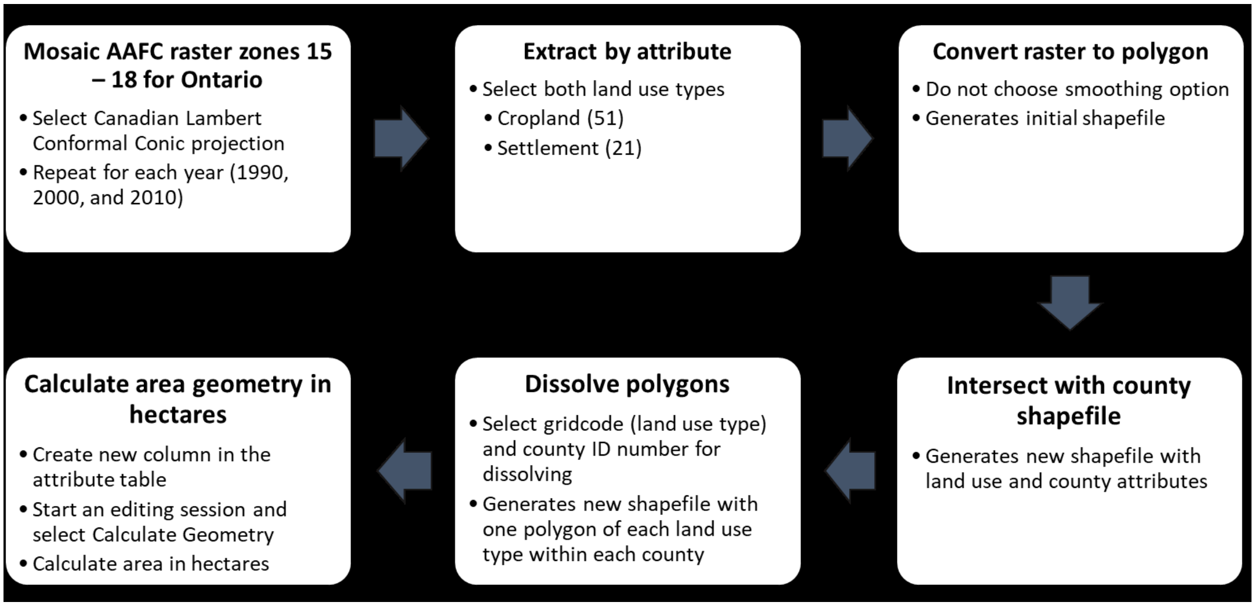
Year	**1961**	**1966**	**1971**	**1976**	**1981**	**1986**	**1991**	**1996**	**2001**	**2006**	**2011**	**2016**
kg P/cow/year	0.22	0.31	0.40	0.49	0.59	0.68	0.77	0.77	0.77	0.77	0.77	0.77

2.2.5 GIS Methods: Converting to Watershed Scale

After NAPI values were calculated for each county, the data were joined with an Ontario county shapefile. The resolution of the county map scale was coarse, especially in northern Ontario where the counties are larger, which made it difficult to delineate spatial differences in P inputs. It was determined that the best way to increase the resolution was to convert the data to watershed scale. Additionally, important hydrological information can be gleaned by examining trends of P inputs within a watershed.

Conversion from county scale to watershed scale involved the spatial redistribution of data. Land use data from Agriculture and Agri-Food Canada (AAFC) were used to assign phosphorus masses in each county to appropriate areas. Agricultural data such as fertilizer application, net livestock inputs, and crop uptake were assigned to land designated as “cropland” (labelled 51 in the dataset), and the detergent and human consumption inputs were assigned to urban areas designated as “settlement” (labelled 21 in the dataset). It was assumed that “cropland” represented all agricultural land, as there was no specific livestock or managed pasture category for Ontario.

The AAFC land use data were used for the years 1990, 2000, and 2010 in zones 15 – 18. The zone rasters were mosaiced into one raster dataset for each year and projected to Canada Lambert Conformal Conic. The land use categories of cropland and settlement were extracted from each of the three AAFC raster datasets by using Extract by Attribute for the desired land use types (51 and 21). The new raster datasets of the extracted land use types were converted to a polygon without smoothing the edges (keeping the square shape of each pixel). The polygons were then intersected with the shapefile of Ontario counties to make a new shapefile that distinguished the amount of each land use type that is within each county. The total area of the land use was calculated for each county polygon in hectares. The attribute table was exported to Excel for use in MATLAB. See below for flowchart of GIS methods for the use of the AAFC land use dataset.



An important step to convert from county scale to watershed scale was to determine the land use area shared between each watershed and each county. These data were calculated in GIS by intersecting the county scale land use shapefile with the watershed shapefile. The output was a new watershed shapefile that contained the county-based land use area within each watershed. The land use area within each watershed was recalculated using Calculate Geometry and exported to Excel.

The mass of P for each category was evenly distributed on the designated land use within each county by dividing the total mass by the corresponding land use area. Afterwards, the total mass of P was reassigned to a sub-watershed level by using the intersected areas calculated for the land use-watershed shapefile. The total mass within the watershed boundaries was normalized within the watershed by dividing by the watershed area. The above can be summarized with the following formula:

Equation 10

$$Total\ Watershed\ P\ (kg) = \sum_{i=county} \frac{M_{AG}}{LUA_i} \times LUA_{watershed(i)_{AG}} + \sum_{i=county} \frac{M_{URB}}{LUA_i} \times LUA_{watershed(i)_{URB}}$$

where M_{AG} and M_{URB} is the total mass (kg P) applied to the corresponding land use area (agricultural area or urban area), LUA_i is the corresponding total land use area (ha) for each county that intersects the watershed, and $LUA_{watershed(i)}$ is the corresponding land use area (ha) that intersects both the watershed and the county. Total Watershed P was then divided by the total watershed area to normalize each watershed to kg/ha for comparison.

2.2.6 Net Cumulative P Inputs

Cumulative inputs were estimated by linearly interpolating the total NAPI for all years between census years. The sum of all years, including years between census years, represented the total net P input into the county landmass over a 55-year period between 1961 and 2016. Initially, the total P from all years for the county was normalized to county area to compare the relative magnitude of P inputs between counties. Total P inputs were also normalized to corresponding land use area, with agricultural inputs such as fertilizer normalized to agricultural land use area, and population inputs such as detergent normalized to urban land use area, to get a finer resolution of P inputs. Cumulative net P inputs were also converted to watershed scale as described above.

2.2.7 Data Sources

Sources of historical data include the Canadian Agricultural Census, Canadian Census of Population, Canadian Fertilizer Institute, Fertilizer Institute of Ontario, the International Plant Nutrition Institute (IPNI), Environment Canada, Ontario's Ministry of Agriculture, Food and Rural Affairs (OMAFRA) Agriculture and Agri-Food Canada (AAFC), and the Ontario Ministry of Natural Resources and Forestry (OMNRF). The most readily available agricultural and population census data were from after the year 1961, and occurred at a frequency of five years, up to 2016. Earlier data were available at a frequency of 10 years from 1901 to 1951 but were less comprehensive. The focus of the study therefore focused on the time period of 1961 to 2016. The full list of data sources for the NAPI analysis are available in Table A-3 (Appendix A).

2.2.8 Obstacles

In some instances, county data were suppressed in the agricultural census. Data suppression occurred if, for a type of crop, there were only one or two small farms in a county, or one large farm in a county. The problem was addressed by comparing the crop area in the county to the total crop area of the region the county was in. Regional data included crop hectare data for all counties in the region, including suppressed counties. The suppressed data points were estimated using the regional proportion of crop area in the county in unsuppressed years, proportionately allocating the remaining area unaccounted for in the region. In other words, the total proportion of land was calculated for that crop in each county relative to the region total, and the average proportion over all years (excluding years with zero growth of that crop) was used to estimate the suppressed cropland area in each county. Suppressed livestock data were estimated using the same method.

The point at which that these data suppressions began to occur was in the 1986 census. The number of suppressed county data in that year and years following was variable and did not follow a pattern. Generally, data suppression was more frequent in the northern region because of the lower density of agricultural activity. Common row crops such as corn and soybeans had very few instances of data suppression because of their widespread planting in Ontario. In total, 15% of used crop data were suppressed, and 4% of livestock data were suppressed. Out of 48 counties, there was an average of 7 suppressed counties per year for crops and an average of 2 suppressed counties per year for livestock. The most suppressed crops were spring wheat, canola, flaxseed, and beans. The most suppressed livestock types were pigs and turkeys.

Additionally, county names, borders, and geographic codes changed over time in Ontario, making it challenging to organize the data to observe temporal trends. To address this, the subdivisions in each county were standardized to their location in 2011, the same year that the county shapefile was produced for GIS. The subdivisions that were amalgamated and transferred to other counties since 1961 were identified and data from those subdivisions were added to their standardized county in 2011 and subtracted from the county they were moved from.

The municipality of Toronto was not included in the agricultural census in 1961, 1966, and 1986 to 2006, and instead the data were amalgamated into the York County data in those years. For this reason, all available Toronto data were similarly amalgamated for all agricultural and urban sources, and land use area.

2.3 Results and Discussion

A time series of maps was developed to demonstrate the annual mass balances of P in Ontario watersheds and urban and agricultural land use areas. This time series showed differences in P inputs during census years, and correlations were drawn between years with notable net P outputs and trends in livestock, crops, and NPS regulations. A positive NAPI value was considered a P surplus, and a negative NAPI value was considered a P removal and would reduce the cumulative surplus value. Regional trends were analyzed to gain a sense of the drivers of spatial variations of P inputs. Cumulative inputs were also examined to highlight areas with significant long-term P surpluses.

2.3.1 Maps

Northern Ontario P inputs changed very little over the past 25 years. Most watersheds in the northern region maintained an annual P input of less than 1 kg P/ha, while some watersheds were completely untouched by anthropogenic influence (Figure 2-6). Southern, western, and central regions had the highest amount of P surpluses, and demonstrated the most observable changes over time, due to intensive agriculture and high population density. The magnitude of P surpluses in southwestern Ontario increased from 1961 to 1981 (Figure 2-7), attributable to intensifying fertilizer use. A decrease in the amount of annual P surplus was observed starting in 1986 and continued until 2016. Blue indicated a decrease in the cumulative surplus removing excess P from previous years; more watersheds had a net removal of phosphorus in later years (Figure 2-6). The year 2006 had the most watersheds with an annual net P removal. All these changes were dominantly influenced by changes in agricultural trends following the "green" revolution due to increasing awareness of the impact of phosphorus on surface waters. Changes in urban influences such as decreasing the allowable concentration of phosphorus in detergents also had an impact, but to a lesser extent.

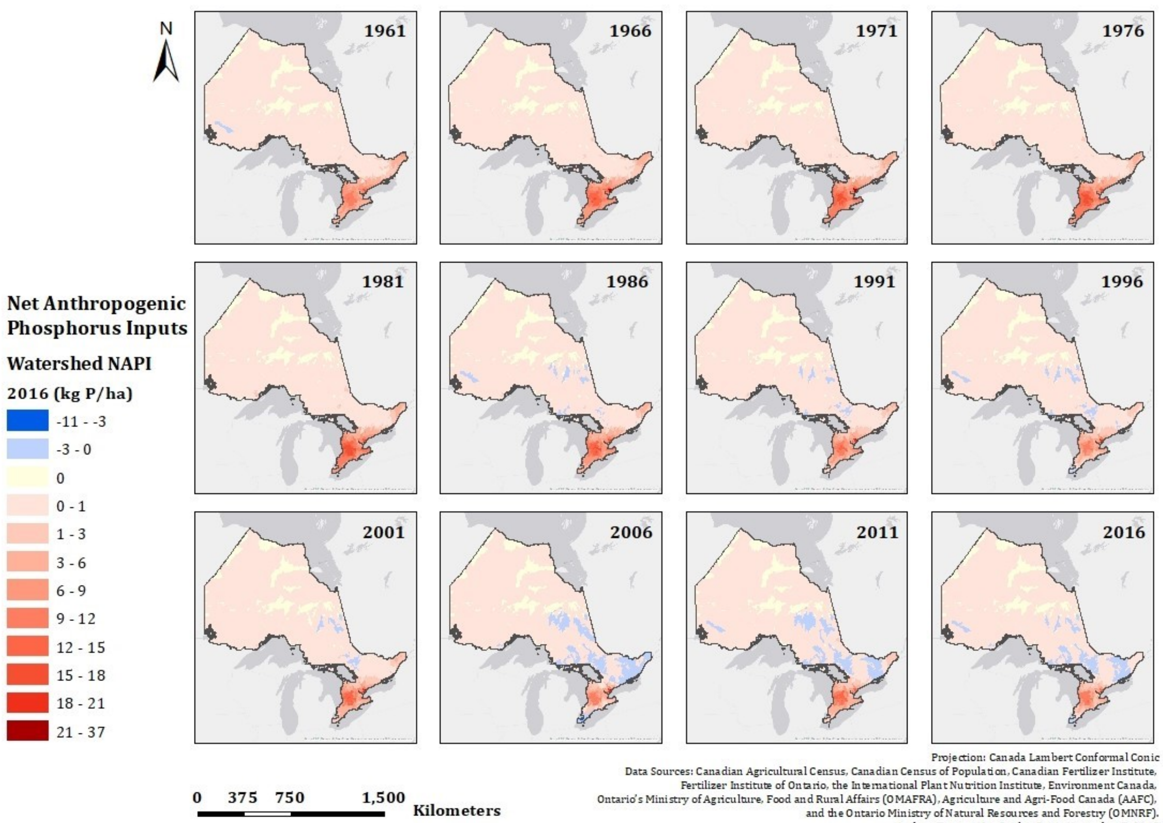


Figure 2-6. Spatial distribution of NAPI in Ontario watersheds.

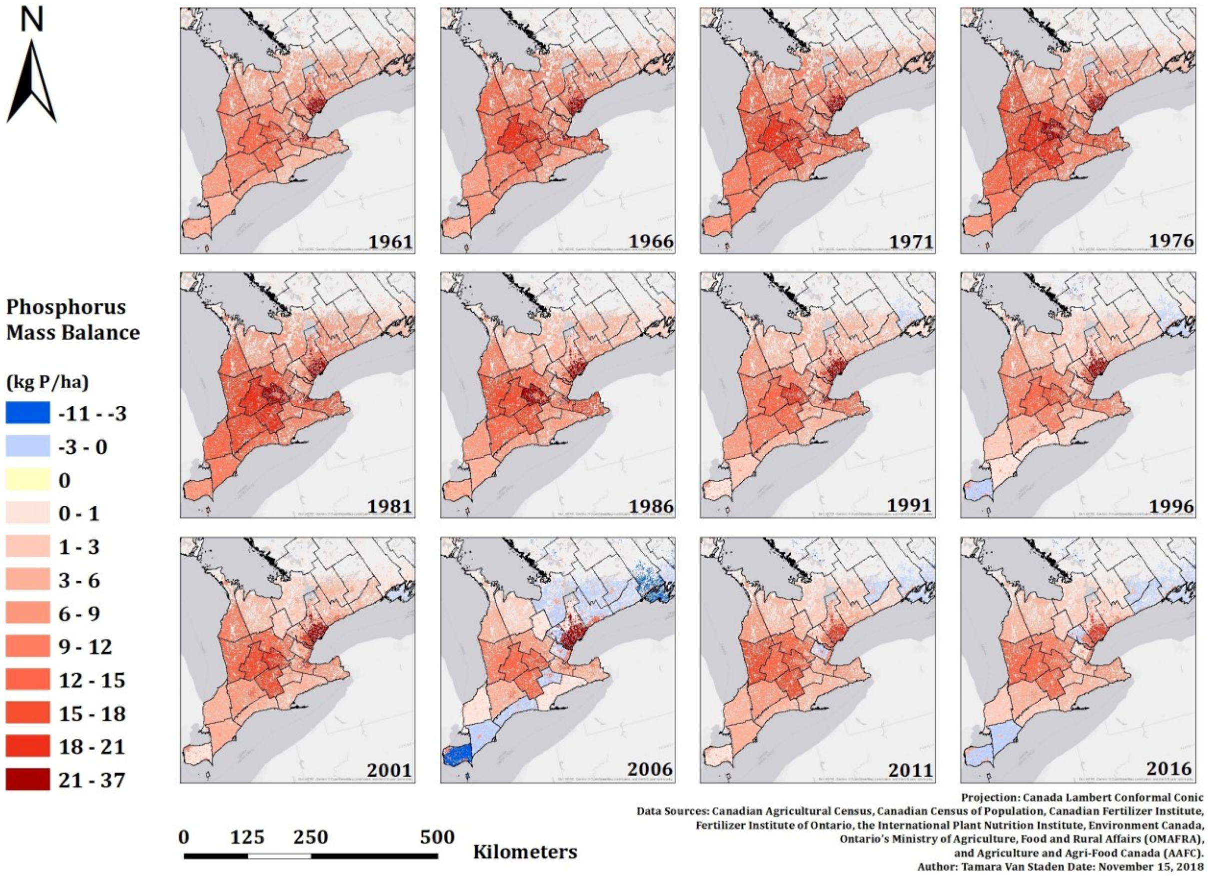


Figure 2-7. NAPI trends in agricultural and urban land use areas in census years of southwestern Ontario.

2.3.2 NAPI Trends

The general trend of P inputs in Ontario overall showed an increase from 1961 to 1976, a slight decrease from 1976 to 1981, and a larger decrease from 1981 to 1996 (Figure 2-8). From 2001 to 2016, there was no specific upward or downward trend, but an oscillation, with 2001 and 2011 having the higher annual surpluses, and 2006 and 2016 having the lower annual surpluses (Figure 2-8). The year 2006 had the lowest overall P surplus across Ontario.

The oscillation in net P inputs from 2001 to 2016 was a result of a culmination of changes in the source of P inputs. Comparing the inputs in the years 2001 and 2006, one major difference was the uptake of P from crops (Figure 2-8). In 2001, P uptake from crops decreased due to a significant decrease in the amount of high yield crops produced (e.g. fodder corn, hay, soybeans) as a result of poor climate conditions (Figure 2-5). In 2006 onward, the crop uptake increased again, lowering the annual surplus. However, fertilizer sales began to increase after 2006 (Figure 2-4), causing the annual P surplus to increase again in 2011 and 2016.

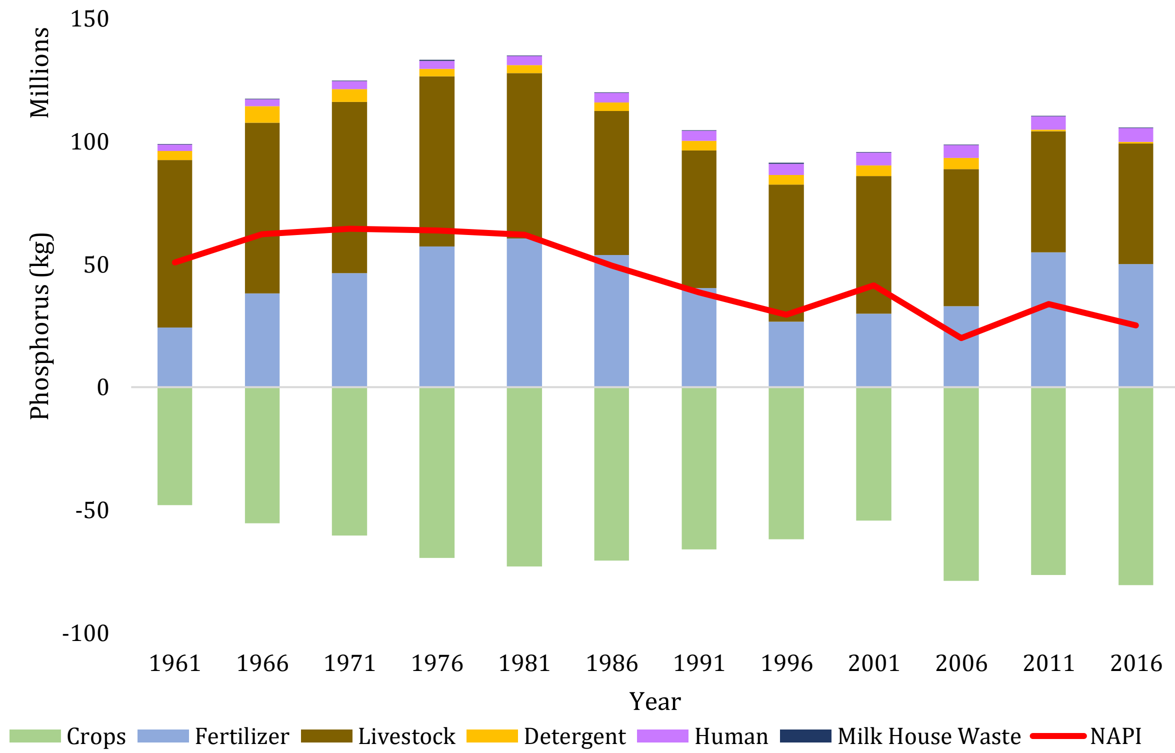


Figure 2-8. Net anthropogenic phosphorus input trends in Ontario.

2.3.3 Agricultural Trends

2.3.3.1 Crops

On average, crop yields in Ontario increased 75% since 1961 (Figure 2-5). Increased crop yields over time contributed to the larger crop uptake of phosphorus in more recent years and played a key role in reducing the surplus phosphorus from soil in later years, which was also observed in MacDonald and Bennett (2009). The type of crop selected by farmers is driven by consumer preferences and economic trends, which has changed over the last few decades (Smith, 2015). There was a significant increase in the mass of soybeans, corn, and wheat grown since the 1970s because of recommended crop rotations and the value of the crops. Notably, these crops have large yields, as well as large increases in yield, when compared with other crops examined (Figure 2-5).

Crop trends in Ontario demonstrated that corn was the largest mass of crop grown with a maximum of 14.1 million tonnes grown in 2016 (both fodder and grain corn), followed by hay, which peaked in 1986 at 7.55 million tonnes. The amount of wheat and soy grown over time has increased significantly, with wheat increasing from 566,000 tonnes in 1961 to 2.85 million tonnes in 2016, an increase of 403%. Soybean mass grown increased from 180,000 tonnes in 1961 to 3.5 million tonnes in 2016, a 1,835% increase. There was a decrease in the amount of oats grown by an order of magnitude, down from over 1.4 million tonnes to just over 100,000 tonnes, a 92% decrease. Barley showed an increasing and then decreasing trend, peaking in 1986 at 780,000 tonnes of barley grown. The amount of tobacco grown decreased by 79%. The mass of canola grown has changed significantly since 1961, beginning in 1986, the amount of canola grown increased nearly 20,000% by 2016. The cause of this enormous increase in canola production was the result of the transition from imported oilseeds to the domestic rapeseed growth in the 1960s and the enhancement of the crop in the 1980s to have more desirable nutrition and flavour characteristics (Casséus, 2009).

Overall trends in Ontario demonstrated an increase in crop mass produced from 1961 to 1981, a decrease from 1981 to 2001, and an increase from 2001 to 2016 (Figure 2-9). The overall linear trend from 1961 to 2016 indicated that grown crop mass increased, which was attributed to increasing crop yields despite cropland area decreasing by nearly 5% since 1921 (Smith, 2015). The decreasing trend from 1981 correlated with decreasing farmland and cropland area in Ontario at the time, likely driven by increased urbanization (Smith, 2015). The significant decrease in crop mass in 2001 was a consequence of the poor climate conditions that year, with notable effects on wheat, corn, and particularly soybeans (Figure 2-10). There were major water deficits in southern Ontario that reduced corn and soy crops yields roughly 50% (Tan and Reynolds, 2003). The summer of 2001 was the driest recorded in 54 years, with a dry spell running from June 25 to August 10 (OMAFRA, 2001). Harsh conditions arose from limited precipitation and high temperatures, which negatively affected crop production. The result of this was limited P uptake and higher NAPI in 2001, despite decreased fertilizer applications that year (Figure 2-8). The implication that climate indirectly affects P removal extends to concerns about climate change, specifically, lower yields increasing the amount of P accumulating in the soil and contributing to legacy P stores.

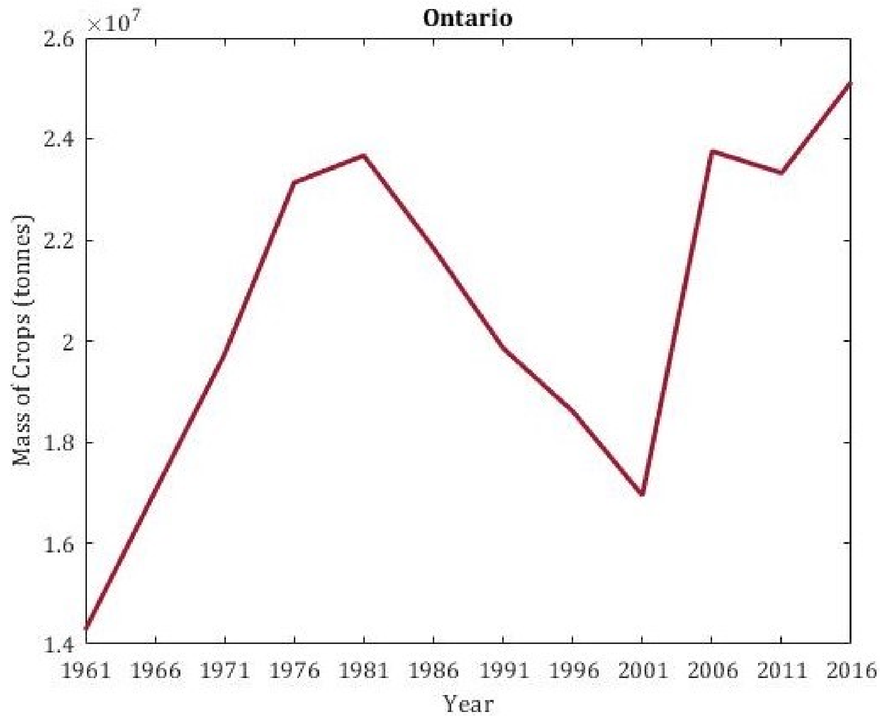


Figure 2-9. Total crop production mass in Ontario census years.

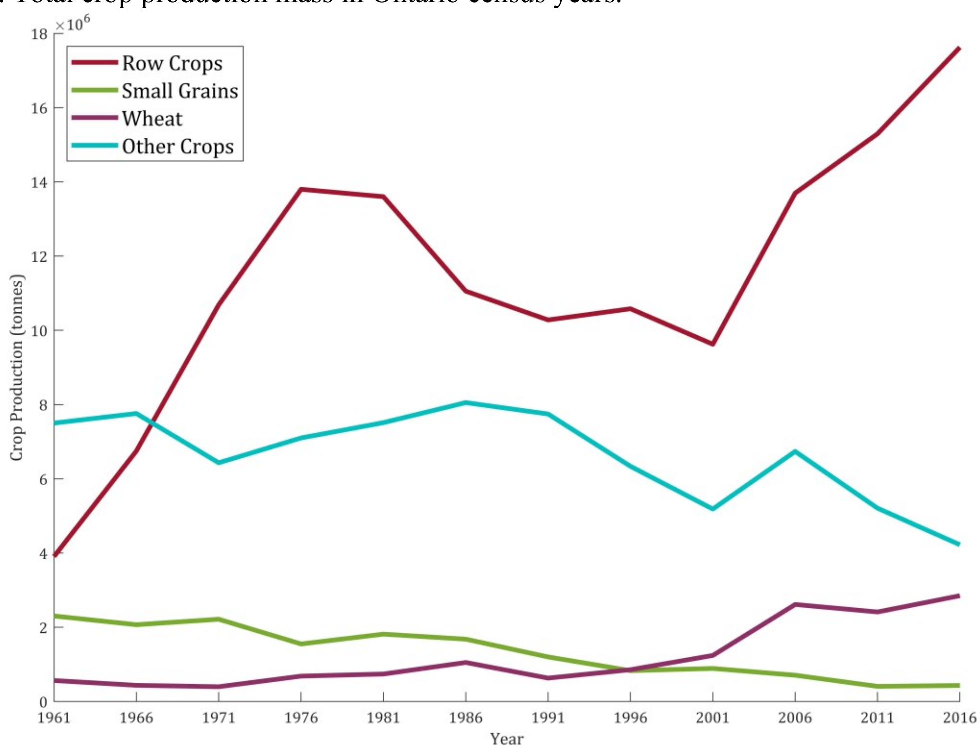


Figure 2-10. Changes in crop production in Ontario census years. Row crops include corn and soybeans; small grains include rye, oats, barley, and mixed grains; and other crops include canola, flaxseed, beans, hay, potatoes, and tobacco.

2.3.3.2 Livestock

There was a decrease in the amount of P attributed to net livestock additions from 1961 to 2016 (Figure 2-8), potentially explained by the changing demographics of livestock inventories. Smith (2015) similarly found that the numbers of different livestock types have changed in Ontario over time. Ontario overall showed a 48% decrease in the number of cattle raised over the past 55 years and an increase in the number of chickens, pigs, and sheep raised (Figure 2-11). Cattle had the largest P input of the livestock examined (Table 2-5), and the significant increase in the number of chickens did not offset the decrease in P caused by reduced cattle numbers, so this decrease resulted in a net reduction in livestock P inputs. Additionally, the mass of manure produced by livestock decreased as a result of reduced cattle numbers despite the increase in numbers of smaller livestock (Smith, 2015).

The reduction in the number of cattle was driven by a number of factors, including increased consumer preference for chicken, larger average size of beef cattle, increased milk production in dairy cows, and reduced consumer demand for dairy (Smith, 2015). Additionally, there were renewed declines in cattle numbers in the 2000s because of the Bovine Spongiform Encephalopathy (BSE) crisis, also known as Mad Cow Disease. It is possible that consumer preferences changed as a result of the BSE crisis. Fewer cattle led to less pastureland in Ontario, and more available cropland (Smith, 2015). Fewer cattle also led to decreased greenhouse gas emissions primarily from the dairy industry (Smith, 2015).

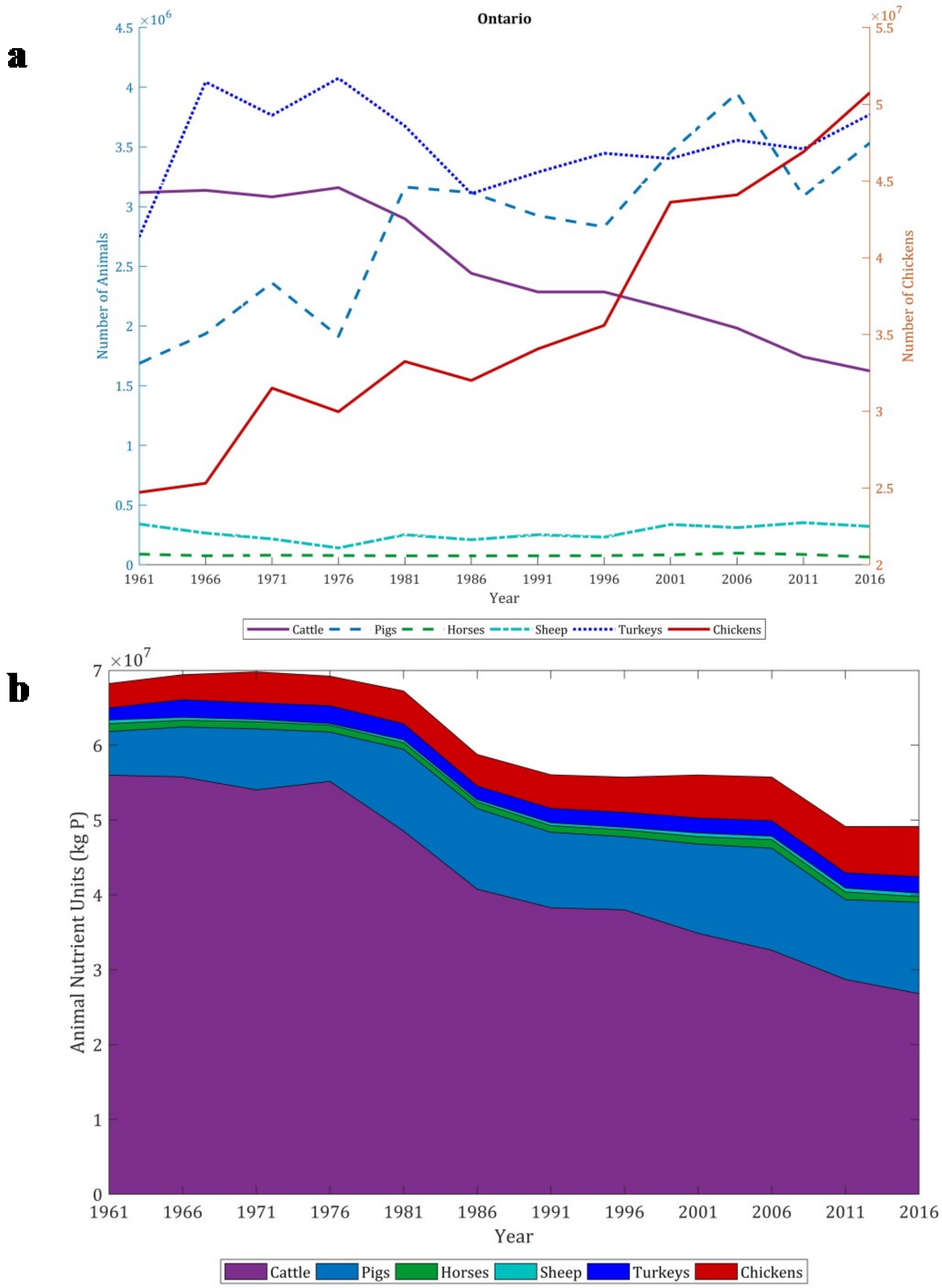


Figure 2-11. Changes in a) the number of livestock, and b) livestock specific nutrient units in Ontario census years.

2.3.3.3 Milkhouse Waste

The consideration of milkhouse waste was a new inclusion in the NAPI model. Milkhouse floor washing runoff often flows directly away from the source without treatment, leading to an additional source of P into watersheds. Quantifying the mass of P from milkhouse wastewater demonstrated that the source is small, but as milkhouse waste inputs were on the same magnitude as detergent contributions in 2011 and 2016, their contribution is still important to consider (Figure 2-8). Cumulatively, milkhouse waste was a notable contributor to P additions in soil and is a potential candidate for mitigation by diversion of wash water for treatment. Research into the P content of detergents used at milkhouses could further the quantification of this source.

2.3.3.4 Fertilizer

Fertilizer sales in early years were largely driven by the "green" revolution after World War II. Commercial fertilizer became immensely popular and reliance on it increased significantly into the 1960s and 70s. The intensification of fertilizer use was observable from the 1960s to the early 1980s (Figure 2-8). Over time, individuals became more aware of their environmental impact, and in Ontario, the HABs in Lake Erie drove individuals and government to act. Government regulations include the Nutrient Management Act of 2002, among many others, which worked to mitigate P runoff through the regulation of fertilizer and manure inputs.

Some watersheds had a net removal of P in 2006 (Figure 2-6), which was driven by a dip in fertilizer sales in 2006 in conjunction with high crop yields (Figure 2-8). However, the current trend of P inputs from fertilizer sales appears to be on an upward trajectory (Figure 2-4). A number of drivers could explain this trend, including changes in fertilizer prices, decreased manure availability as a result of fewer cattle, fertilization needs of preferred crops, decreases in available P in soil, more available land in the northern region, and warmer climates leading to longer growing seasons. Corn and soy row crops as emerging crops grown in Ontario require higher P application rates than other crops grown in Ontario, driving more fertilizer P sales and application. Ontario recommendations suggest that 30ppm of Olsen P is the upper tier for corn, at which P fertilizer application would only achieve minimal crop yield increases (Brown et al., 2017; Munroe et al., 2018). Additionally, fertilizer prices are considerably impacted by global markets and are tied to natural gas prices (Bucknell et al., 2016), therefore, fertilizer prices have the potential to be unstable an can increase in times of economic uncertainty. Further economic analysis may refine our understanding of pricing trends and purchasing behaviour.

The specific driver of the recent upward fertilizer trend is unclear; however, it is speculated that soil P tests were showing lower available P in soil than was commonly observed as a result of nutrient management and an emerging cropland P deficit (International Plant Nutrition Institute, 2013b). In 2015, 35% of Olsen P results in Ontario tested for >38 ppm, down from 50 % in 2001

(International Plant Nutrition Institute, 2013b; LimnoTech, 2017). Additionally in 2010, 31% of soils tested below the critical Olsen P level of 20 ppm, a larger proportion than in previous years (International Plant Nutrition Institute, 2013b). Consequently, more fertilizer than usual might have been applied to the soil to counteract the lower soil P test results to ensure desired crop yields. However, it is important to consider that the soil P test results do not represent the magnitude of legacy P below the first 15 cm measured. Additionally, stores of P can become bioavailable with changing redox and pH conditions.

2.3.4 Urban Trends

2.3.4.1 Detergent

Detergent P was successfully regulated in the 1970s, significantly reducing the amount of P in laundry detergent (Figure 2-3). It should be noted that the increase in amount of dishwashing detergent per capita throughout the 70s was primarily a result of increased use of dishwashers from the 1950s to the 1990s (Han and Bosch, 2012). Detergent P was further reduced in 2010 with regulations from the Government of Canada which affected all detergent products within and being transported into Canada (Government of Canada, 2010). The influence of these regulations were seen in urban areas in 2011 and 2016 census years, especially in the Toronto area, where, despite continuous population growth, the P inputs from urban areas decreased from 29 kg P/ha to 18 kg P/ha (Figure 2-7).

2.3.4.2 Human Consumption

Per capita human consumption remained relatively stable over the years, although the amount of P available decreased slightly in the 2000s, possibly driven by decreased meat consumption. Purchasing trends in Ontario showed that people have been proportionately spending less money on meat products in the 2000s when compared to the 1960s (Figure 2-12). According to Statistics Canada, households went from spending 27 cents to 20 cents on meat for every dollar spent on food from 1982 to 2001, taking inflation into account (Statistics Canada, 2003). Additionally, the proportion of households that reported purchasing beef decreased by 5% between 1982 and 2001. This could be a result of the BSE crisis, changing prices, a change in preferences, or changing diets.

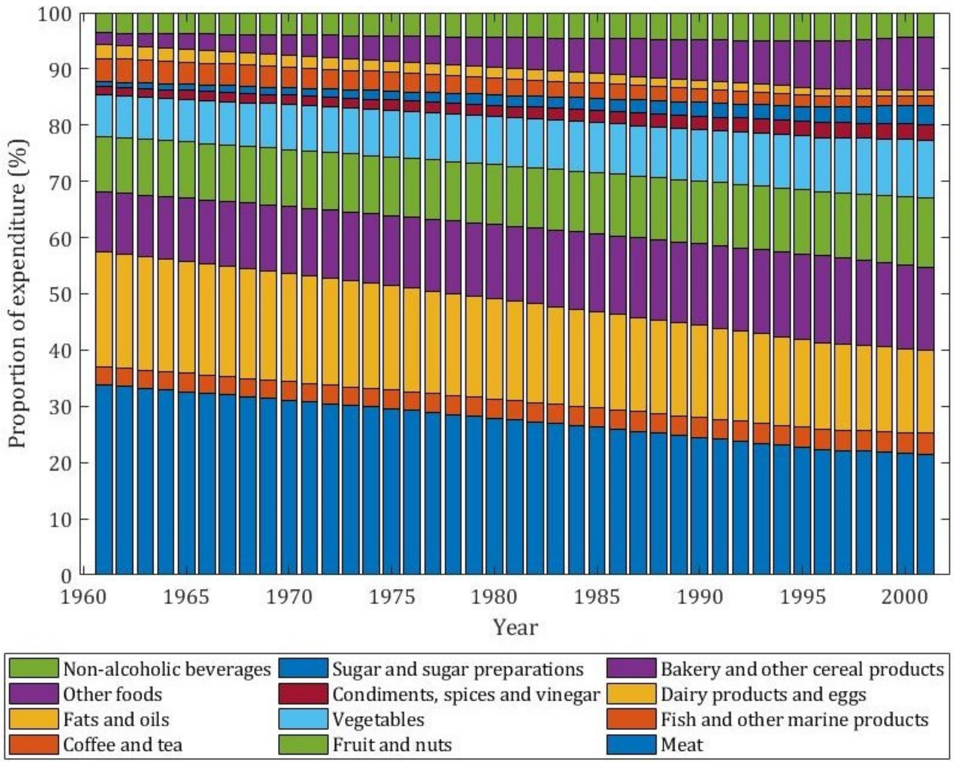


Figure 2-12. Fractional expenditure on food in Ontario (Statistics Canada, 2003).

2.3.5 Regional Trends

2.3.5.1 Regional NAPI Component Trends

All Ontario regions demonstrated a generally decreasing net P input over time; however, southern and western Ontario were the largest contributors, even when normalized to land use area (Figure 2-13). Northern Ontario had the smallest decrease in NAPI compared to all other regions. Southern and western Ontario were the main contributors to the total P inputs in Ontario overall, which could be a result of higher density of agriculture within their agricultural land use area, including larger livestock operations.

The oscillating pattern in later years was driven by significantly changing crop yields in those years, for example, 2001 had a low crop yield, thus reducing P removal and driving NAPI upward. This trend was more prominent in southern, western, and eastern Ontario, agriculturally dominated regions. In earlier years, central, eastern, and northern Ontario followed a similar downward trend driven by decreasing livestock and increasing crop production.

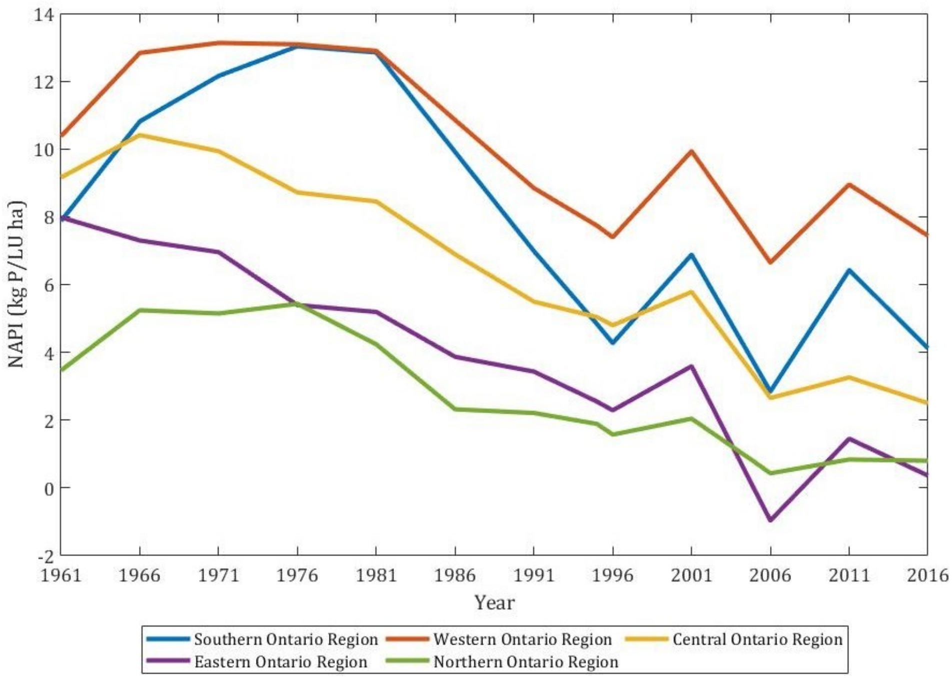


Figure 2-13. Regional NAPI trends in Ontario normalized to agricultural and urban land use area.

The contribution of P from livestock differed significantly between southwestern regions and other regions. In the eastern, central, and northern regions, as well as most counties, there was a decrease in the number of cattle raised. However, southern and western Ontario counties had a stable, or even increasing number of cattle, driving their higher P inputs in later years. Comparing regional net P inputs from livestock normalized to land use area, the southern and western regions had the largest inputs in more recent years, and the smallest downward slopes, -0.04 kg P/ha/year (r^2 = 0.65) and -0.03 kg P/ha/year (r^2 = 0.30) respectively, an order of magnitude smaller than all other regions (Figure 2-14).

Waterloo Region, within western Ontario region, demonstrated an increase in the net contribution of P from livestock, and had the largest normalized contribution of P from livestock from 1996 onwards (Figure 2-15). Haliburton county had the largest contribution previously but experienced a significant decrease in livestock contributions throughout the late 1900s. The number of cattle in Waterloo increased by approximately 16,000 individuals from 1961 to 2016. The increasing trend in P contributions from livestock was observed in five counties in southwestern Ontario (Figure 2-15). All other counties had decreasing cattle numbers, and therefore less manure and less P inputs from livestock over time.

The regional livestock trends were further examined by grouping the regions that had similar trends and comparing the P contributions from the different livestock types: northern, eastern, and central Ontario had steep downward trends, and southern and western Ontario had shallow downward trends (Figure 2-16). The impacts of decreasing cattle numbers were observed in the north, east, and central Ontario regions when compared with southern and western regions (Figure 2-16). Per agricultural land use hectare, the northeastern regions had a much larger decreases in P contributions from livestock, with fewer cattle inputs dominantly driving the trend downwards. Livestock P from southwestern Ontario remained relatively constant, with cattle inputs decreasing relatively moderately, and pig and chicken inputs increasing in tandem (Figure 2-16).

The density of intensive livestock operations (ILOs) in southwestern Ontario could explain the livestock input trends. There are now fewer livestock farms in Ontario when compared to 1961, but each farm has larger livestock numbers, indicating a trend towards concentrated livestock raising (Beaulieu et al., 2001). More and more farms in these regions developed concentrated operations to keep up with demand. In 1996, the highest densities of animal units, as defined by OMAFRA, were found in the counties listed in Figure 2-15, with some extension into Bruce county and Grey county (Beaulieu et al., 2001). The impact of these high livestock concentrations on phosphorus inputs can be seen in these counties throughout the county scale mass balance time series (Figure 2-7).

The high density of livestock in southwestern Ontario has been well documented and is recognized by government and rural residents (Brückmann, 2000). Efforts have been made to reduce the impacts of livestock operations of all sizes through the implementation of manure-related CPs such as manure storage and manure application plans (Woyzbun, 2012). There was a strong correlation observed between counties with high manure volumes per hectare and the number of manure management related CPs implemented in those counties during the period of 2005 to 2010 (Woyzbun, 2012).

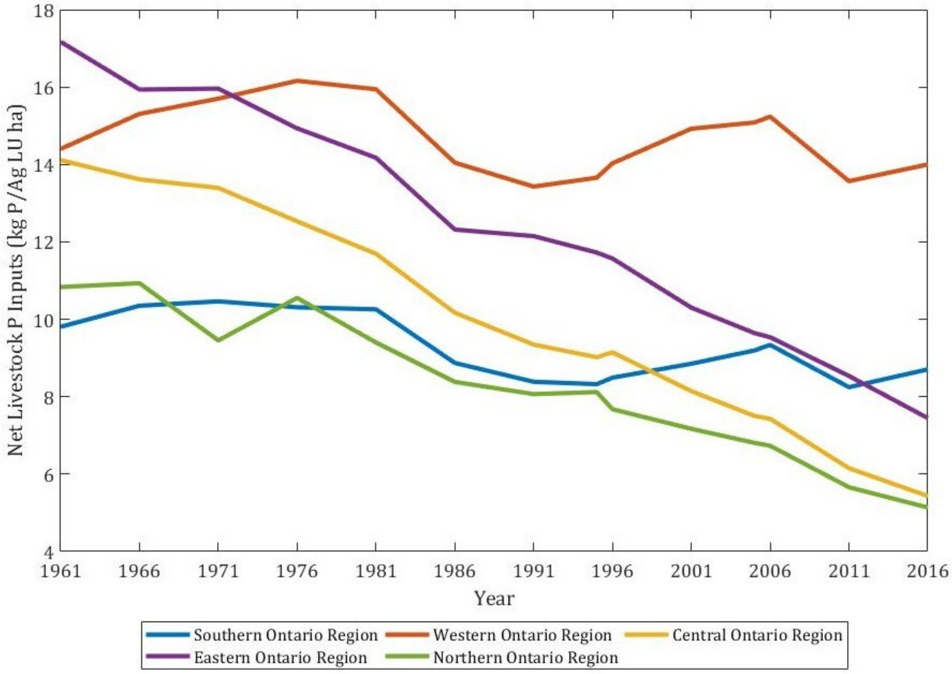


Figure 2-14. Regional livestock P input trends in Ontario normalized to agricultural land use area.

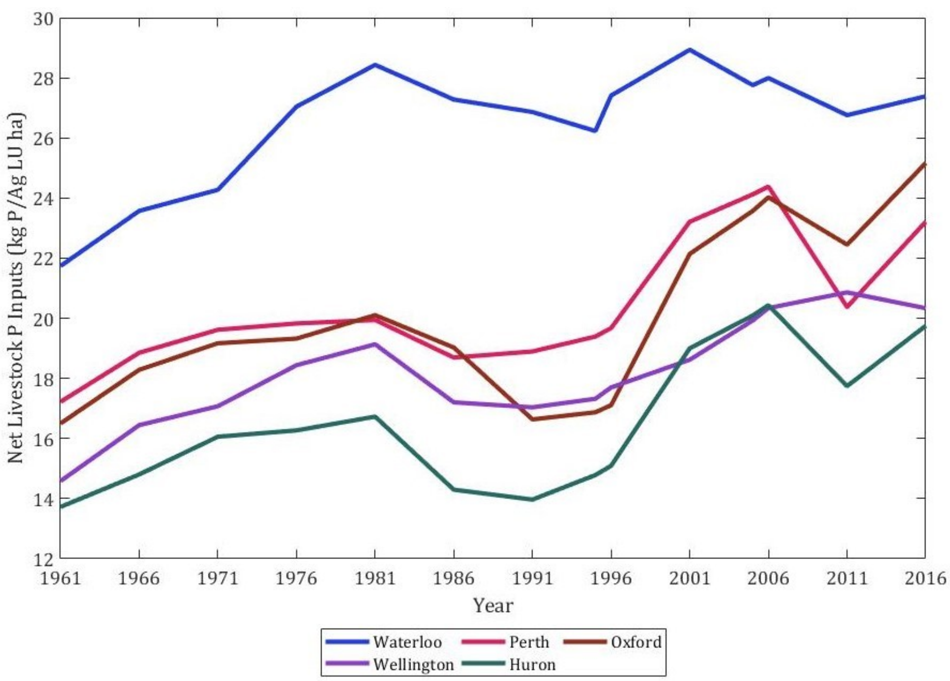


Figure 2-15. Counties with increasing net livestock P input trends in southern (Oxford) and western (Waterloo, Perth, Wellington, and Huron) Ontario, normalized to agricultural land use area.

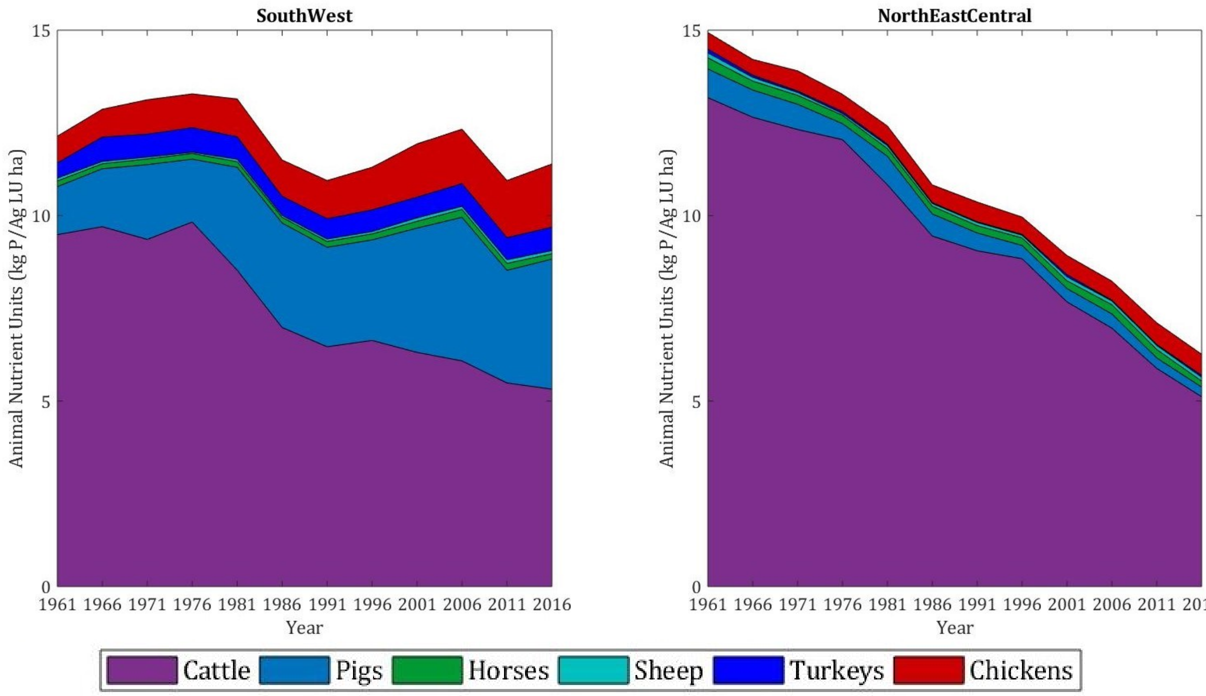


Figure 2-16. Kilograms of phosphorus from each livestock type from southern and western Ontario compared to northern, eastern, and central Ontario, normalized to agricultural land use area.

Fertilizer application magnitudes differed between agricultural regions. Southern Ontario had the highest use of fertilizer in both mass (Figure 2-17a) and normalized use (Figure 2-17b), followed by western Ontario. The assumption that fertilizer was applied at the same rate across the province meant that regions with more fertilized area were assigned more fertilizer, suggesting that southern and western Ontario had more fertilized area per unit of agricultural land. The high volume of fertilizer P use throughout the 70s led to higher NAPI values, especially in southern and western regions (Figure 2-13). Note that the total mass of fertilizer represents annual sales data (Figure 2-17a), so specific peaks were not reflected in the regional curves, which were estimated every 5 years.

a

b

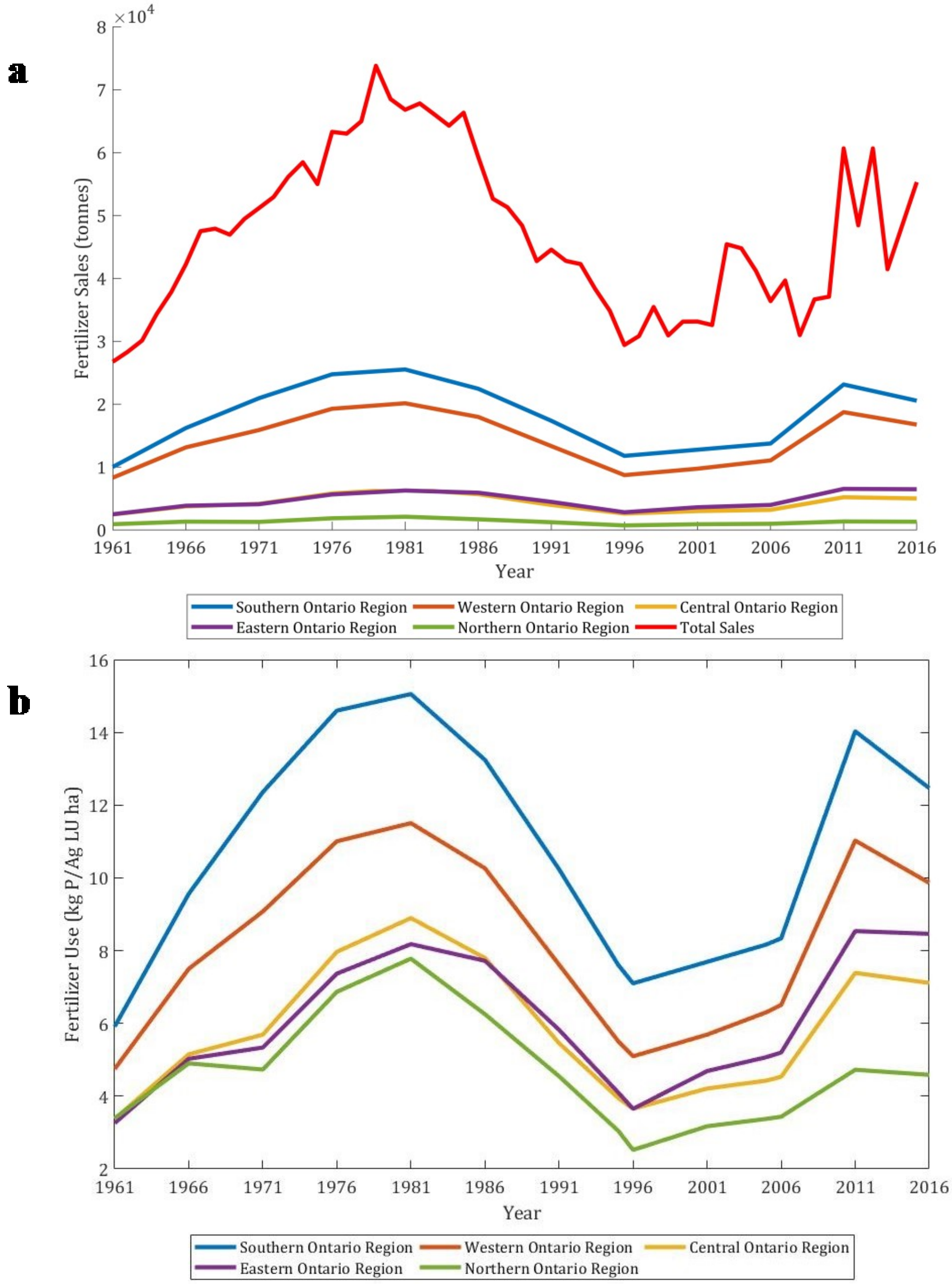


Figure 2-17. (a) Regional fertilizer trends as total P application in tonnes (estimated every five years) and total fertilizer sales in the province (reported annually); and (b) P application normalized to agricultural land use area in kg/ha in census years.

2.3.5.2 Regional NFFI Trends

Net Food and Feed Inputs considered the consumption needs of humans and livestock, classified as consumers, within each region and compared that with the crop and livestock production, classified as food available. Estimated NFFI in each region gave an indication of whether the region required P imports to meet the needs of the consumers, or if excess was produced, and the region exported the extra P that year. A positive NFFI indicates that the system is importing P, and a negative NFFI indicates that the system is exporting P.

All regions started as P importers, and there was a movement towards P export in all regions, indicated by a downward slope. Western and central Ontario maintained their status as P importers over time, but for different reasons. Western Ontario, as discussed, has a significant livestock population, and therefore would have imported a large amount of P in the form of feed. Central Ontario, however, had a high human population density, and so P imports would have come in the form of food. This was apparent from the larger total livestock P inputs and human consumption P inputs normalized to land use area for western Ontario and central Ontario respectively (Figure 2-18). Eastern and northern Ontario regions moved towards a net export of P due to the decreasing numbers of livestock in those regions, and therefore decreased consumption needs (Figure 2-18). Southern Ontario moved towards a net export of P in 2006 as a result of the increasing crop production trend, exceeding both human and animal P consumption needs (Figure 2-18). All regions demonstrated an upward spike in 2001, leading to P export, as a result of low crop yields that year.

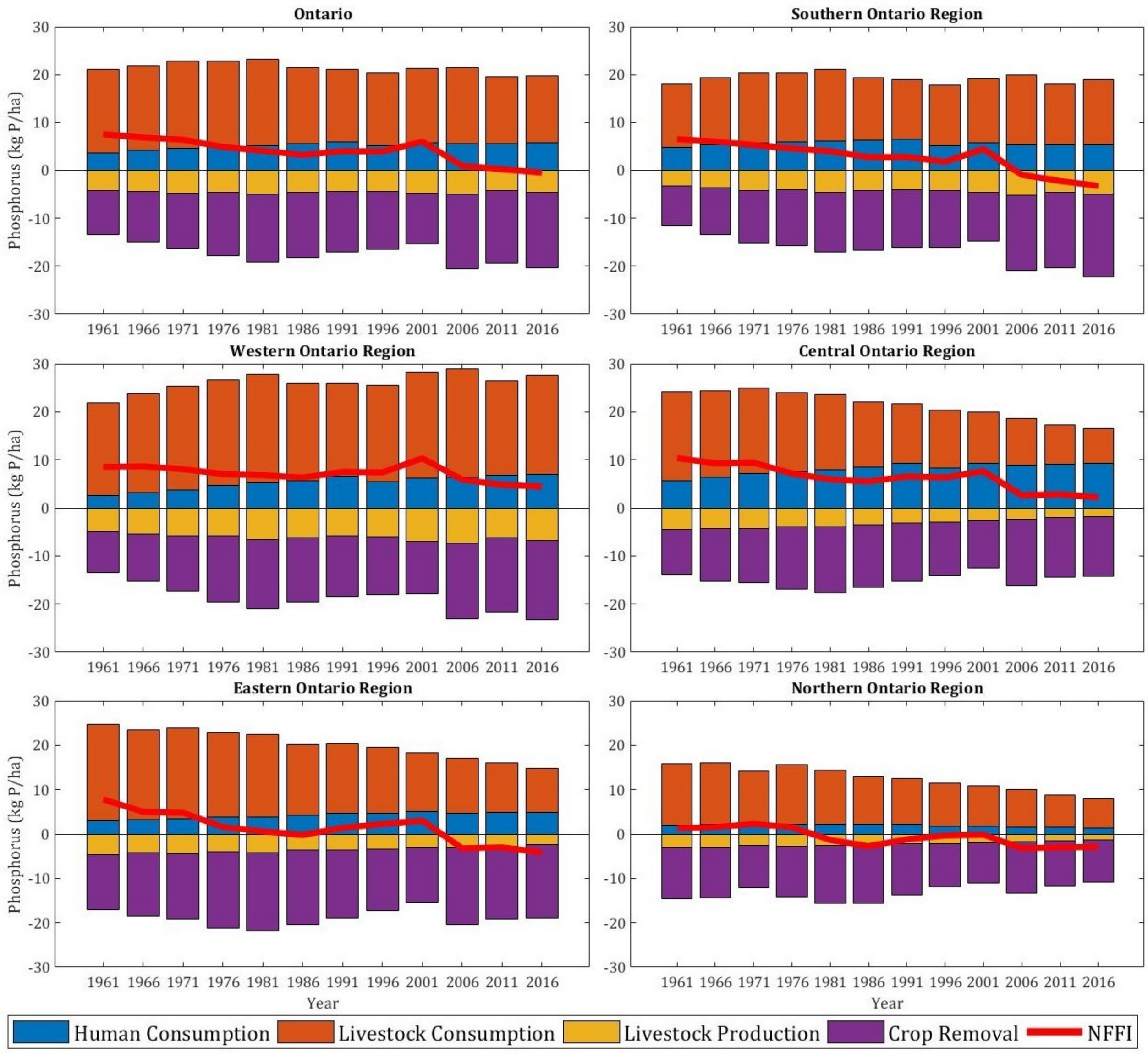


Figure 2-18. Net food and feed import (NFFI) data for Ontario regions and Ontario overall. Positive NFFI represents a net import of P, and negative NFFI represents a net export of P.

2.3.6 Cumulative

The area with the largest anthropogenic cumulative P input from the year 1961 to 2016 was southern Ontario (Figure 2-19). The dense population in this region, combined with the significant amount of agricultural area, provided a long-term source of P into Ontario watersheds. Notably, P from human consumption and detergent would have passed through wastewater treatment plants where a significant amount of P would have been removed and transported to landfill (Samson, 2019). However, the non-point agricultural sources have the potential to accumulate in soils and groundwater. Both point and non-point P sources were included in a cumulative map to delineate areas with high P input density (Figure 2-19). Comparing the watershed cumulative P mass balance map with the Ontario land use map, there was a correlation between dense agricultural land and higher cumulative P inputs. The watersheds with zero kg P/ha highlight areas with no inhabitants, and relatively little anthropogenic influence. These areas may be impacted by atmospheric deposition of P, which was not included in the model.

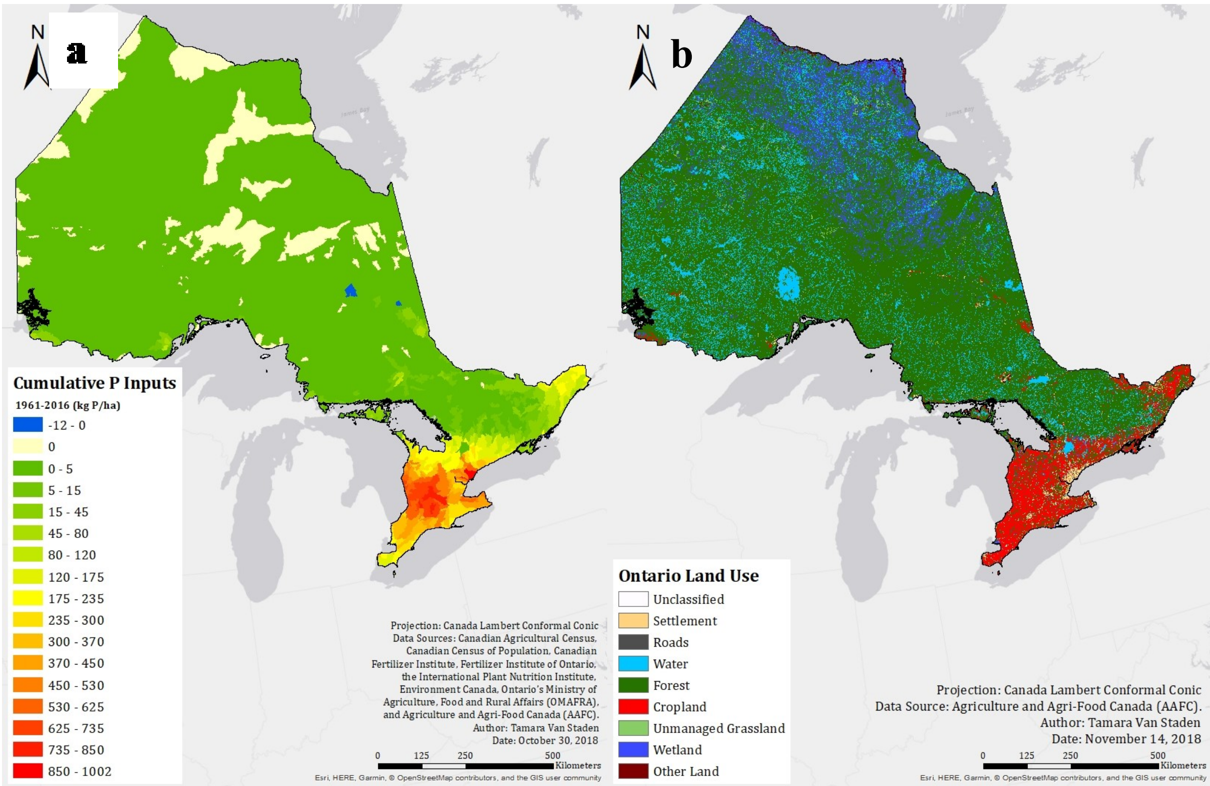


Figure 2-19. Comparison between phosphorus inputs and land use in Ontario. a) Cumulative P inputs in Ontario watersheds from 1961 to 2016; b) Land use in Ontario in 2010.

The agricultural P cumulative map included only non-point (i.e. agricultural) source P inputs from the NAPI model and represents a long-term soil P balance (Figure 2-20). The higher density of agricultural land in southern and western Ontario had a larger P surplus relative to other agricultural areas. The proximity of these P surpluses to the shallow waters of Lake Erie increased the vulnerability of P losses through nearby streams, especially considering the potential P loading to the Thames River and the Grand River. Realized vulnerability to P losses depends on other characteristics such as soil texture, slope, and precipitation, which affect erosion potential and particulate P losses, and pH, soil iron content, and humic acid presence, which affect the fate and transport of dissolved reactive P. Northern watersheds were minimally impacted, and the watersheds with zero kg P/ha highlight areas with no agriculture.

A few watersheds in northern Ontario had a net removal of agricultural P over the past 55 years (Figure 2-20). A combination of factors could have driven this, including land use. Crop uptake was the only P removal considered in the model, and perhaps in these watershed areas the agricultural practices and the increased yields in crops in later years offset the historical fertilizer inputs. Land use change could have driven the P removal if agriculture extended into this area in later years when P inputs from fertilizers had decreased and removal of P by crops was more efficient. It is possible that crop yields in the northern region were lower than the provincial average, meaning that the cumulative P value would increase from reduced crop P removal. In this case, we could see the watersheds with a net removal move towards a net balance or a small net surplus of P.

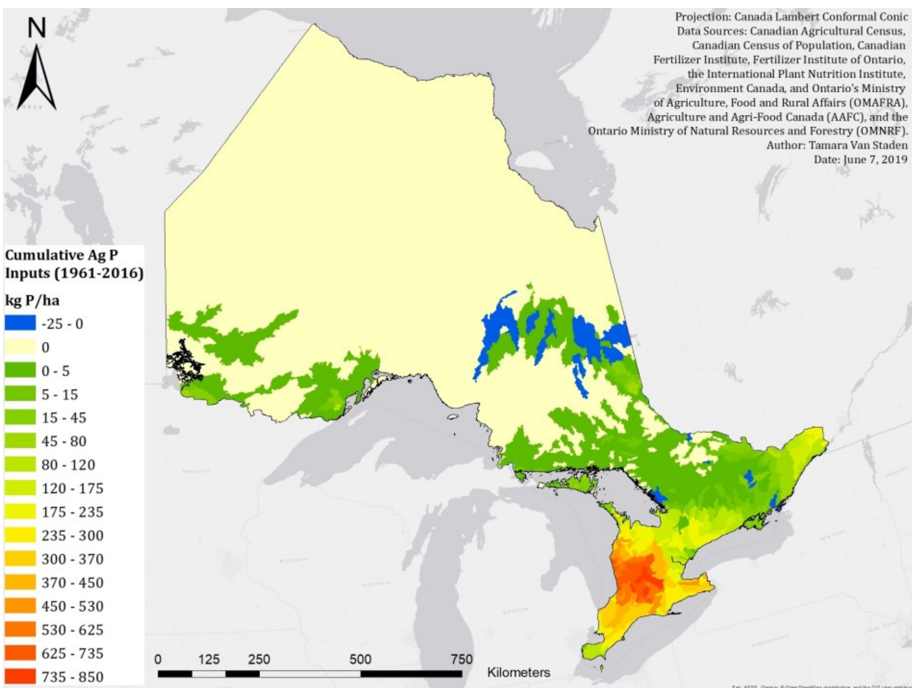


Figure 2-20. Cumulative agricultural P inputs in Ontario watersheds from 1961 to 2016.

Cumulative land use maps showed that York and Toronto had the largest cumulative urban input (Figure 2-21a) and the Region of Waterloo had the largest cumulative agricultural input (Figure 2-21b). York and Toronto together represent the most populated area in Ontario, therefore urban inputs were expected to be high. The high agriculture inputs in Waterloo region compared to other counties was driven by the high inputs from manure. Waterloo region had the highest cumulative input from manure, 1486 kg P/ha, over 300 more kg/ha than the next highest contributor, Haliburton county. However, cumulative crop production in Waterloo region was only the third highest at 1009 kg P/ha, below Thunder Bay and Haliburton county.

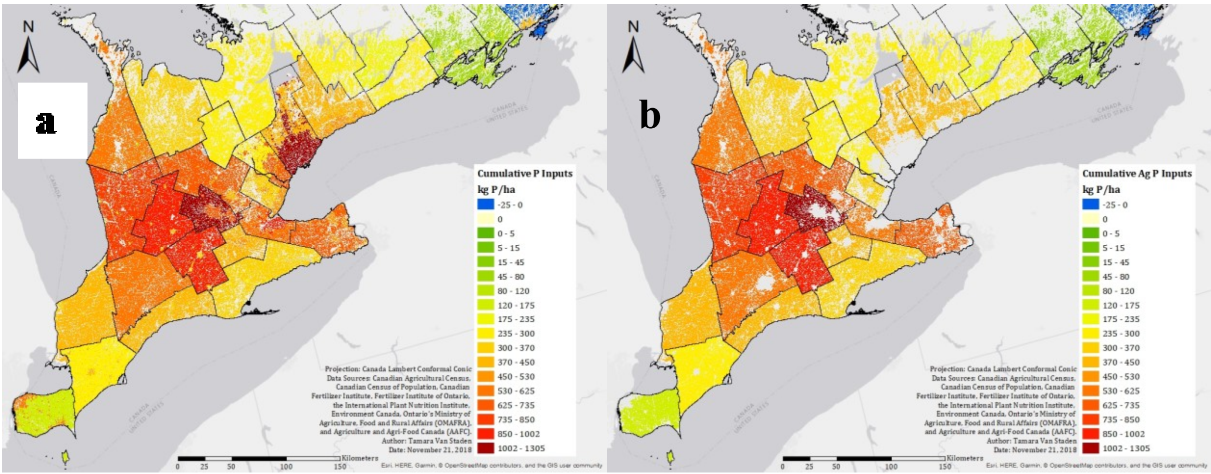


Figure 2-21. Cumulative land use P inputs in southern Ontario: a) Cumulative inputs in urban areas and agricultural areas; b) Cumulative inputs in agricultural areas only.

2.3.7 Sources of Error

Considering the distribution of areas with a net removal of P in census years, it is possible that there was a misrepresentation of manure application locations. In areas such as Essex, where there are few ILOs and few livestock overall, it is possible that some manure trading between counties takes place, but this is unlikely. Manure trading data were not available, or did not exist, however, estimated regional manure totals were representative of manure totals in that region. The assumption that manure remained on fields within the same county was reasonable as most manure types are difficult to transport far distances. Cattle manure usually stays within the watershed it was produced in, one study finding that 94% of fields with manure application received the manure from the same watershed. The same study found that the median distance of manure transport away from large animal operations was 5.7 km (Long et al., 2018). It should be noted that chicken manure is easier to transport due to its lower moisture content, it could be transported more than 5.7 km away and exported to counties such as Essex. However, there are limitations to transport distances; Long et al., 2018 found the maximum distance to be 33 km.

For simplicity, an even distribution of P inputs on land use areas was used. However, this approach could be misrepresenting P input locations in urban areas especially, without accounting for population density factors. Agriculturally, the distribution of crops varied spatially within each county and changed each year. Different crops have different P needs and would require different rates of fertilization. Future analyses would benefit from using the crop census data to weight the distribution of fertilizer (e.g. counties with a higher proportion of corn would be assigned more fertilizer). Managed pastureland was also not differentiated within the southern Ontario land use data used. Therefore, manure and fertilizer applications were not distinguished between these areas, potentially skewing the results as pastureland area has been decreasing over the past several decades. Some land use data was available from Agriculture and Agri-Food Canada that showed the spatial location of crop types and pastureland within most of Ontario, however, these data only dated back to 2009, and would only be available for two of the twelve census years examined. A single year analysis could be conducted to get a better spatial resolution of P input distribution. A finer focus using the specific fertilization rates for the different crops and county scale yields could help refine these estimated.

Atmospheric deposition of P was not utilized in the mass balance as there was no spatial variation in available atmospheric deposition data. It is also difficult to reliably measure wet and dry atmospheric P deposition and even so, there is a significant amount of uncertainty associated with reported results (Maccoux et al. 2016). A single value of 0.2 kg/ha/year was reported for southern Ontario by Winter and Duthie (2000), atmospheric deposition data from 2003 to 2013 were also available and had an average value of 0.19 kg/ha/year (Bocaniov, personal communication). The reported atmospheric values represented less than 5% of the annual anthropogenic inputs in southwestern agricultural regions but represented up to 57% of the inputs in the northern agricultural region. For areas with little anthropogenic influence, such as the watersheds in northern Ontario (Figure 2-19 and 2-20), atmospheric deposition may represent the majority of anthropogenic inputs. Further spatiotemporal studies of atmospheric P deposition could fill this knowledge gap and contribute to a more accurate quantification of P legacies in Ontario.

2.3.8 Uncertainty

A single province-wide value was available to represent temporal trends for yields, introducing uncertainty as yields were likely different between the northern and southern regions especially. Yield values were only available on a provincial scale prior to 2004 and the actual regional differences in yields were available for only three of the census years (2006, 2011, and 2016). There was no consistent trend that showed any region had the highest yield for all crop types, however, southern and western Ontario had higher yields more often in all three census years, and for prolific crops such as corn and soy, but it remains uncertain for earlier census years. Central Ontario consistently had crop yields that were lower than the Ontario average, meaning that NAPI was likely higher in this region with less crop P uptake. Northern Ontario had yields higher than southern Ontario for oats for the three census years, and canola, beans, and spring wheat in some years. However, the influence of these crops is less significant than corn and hay crops which are produced in larger volume in northern Ontario. Despite this, high NAPI values in earlier census years driven by fertilizer use (Figure 2-17b), as well as the potential underestimation of fertilizer use in southwestern Ontario, left southern and western Ontario as the significant cumulative NAPI contributors.

Uncertainty associated with yield factors, when averaged over available crop yield data in the three census years, introduced an 18% error of overestimated P removals by crops in northern Ontario (Table 2-7). Central and eastern Ontario had overestimated crop P removals of 15% and 6% respectively, and southern and western Ontario had underestimated crop P removals of 7% and 3% respectively (Table 2-7). In conjunction, considering that northern, eastern, and central Ontario had overestimated fertilizer P inputs (Table 2-2 and Table 2-7), the uncertainties associated with crop yields and fertilizer distribution balanced each other to some extent. Similarly, southern Ontario had underestimated fertilizer P inputs, which could have balanced the underestimated P removal by crops. Western Ontario was the exception, with underestimated P removals and overestimated fertilizer P inputs, meaning that NAPI could have been much lower than reported in this region (Table 2-7). However, it is unlikely that this would offset the impact of the ILOs; although western Ontario would likely not require as much fertilizer because of the manure from the livestock farms.

Table 2-7. Estimated uncertainty associated with fertilizer and crop yield estimations. Negative average differences indicate that the potential value was lower than the reported value, and positive values indicate that the potential value was higher. For fertilizer, a negative value means that NAPI would decrease, and for crops, NAPI would increase.

Region	Fertilizer		Crops		Regional NAPI Difference	Ontario NAPI Difference
	Average Difference (%)	Average Deviation from NAPI (%)	Average Difference (%)	Average Deviation from NAPI (%)	(%)	(%)
South	11.72	16.46	7.27	-12.30	4.16	1.44
West	-5.13	-3.90	2.94	-6.57	-10.47	-4.86
Central	-1.40	-1.26	-15.40	30.74	29.49	3.53
East	-2.77	-6.06	-6.48	28.15	22.09	1.70
North	-2.42	-3.95	-18.09	69.65	65.70	1.22

Combined, the crop and fertilizer uncertainties increased the potential NAPI in each region except for western Ontario (Figure 2-22). Western Ontario's potential NAPI decreased because of the combination of less fertilizer applied in the region, and higher crop yields. Northern Ontario had the largest percentage increase; however, the magnitude of the increase was small due to the low NAPI in the region (Figure 2-22). Overall, NAPI was not altered significantly (< 5%) considering only fertilizer and crop uncertainties.

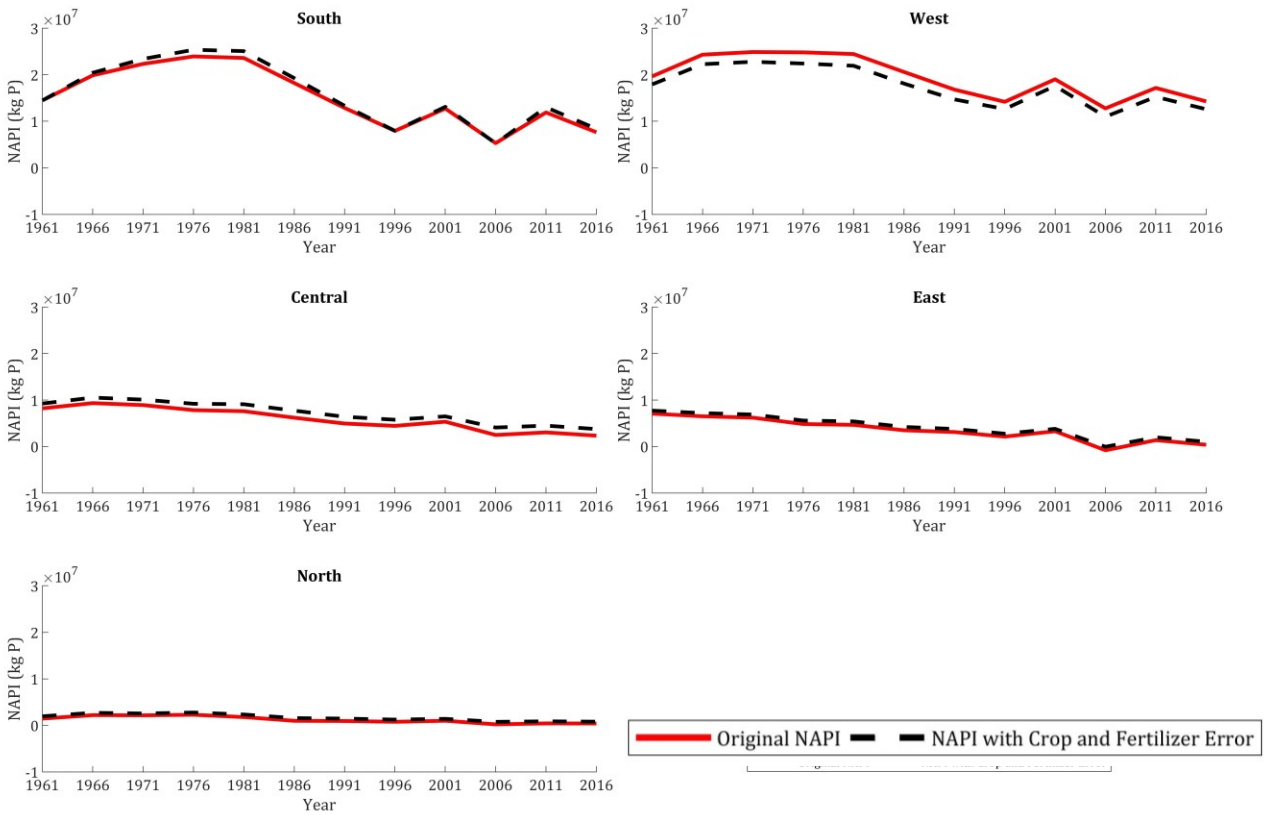


Figure 2-22. Potential deviation from NAPI in each agricultural region considering the uncertainties associated with crop yields and regional fertilizer distribution.

In addition to uncertainties associated with fertilizer and crop yields, there was uncertainty associated with the net livestock inputs. A single value for each livestock type was used to represent intake and excretion (Table 2-5). These values may vary over time and in different countries, however, temporal variations were not available in similar studies (van Bochove et al., 2010; Hofmann and Beaulieu, 2006; Kellogg et al., 2000; Lander et al., 1998). Instead, a range of values were taken from these studies and used to set high and low uncertainty bounds (Table 2-8). The average value was taken from the entire range of values found. No alternative values were found for horses or sheep.

Cattle had the most significant impact on livestock P inputs due to the magnitude of P per head. Most values from other literature for cattle types were lower than the values used (Table 2-8). Only calves and dairy cows had values from other literature that were higher than the values used (Table 2-8). The influence of the lower bound would be larger in areas with more cattle, which can be seen in southern and western Ontario where there are denser livestock populations (Figure 2-23). Most significantly, southern and western Ontario had a net -18% and -35% influence on Ontario NAPI respectively when considering the lower bound of values. This is possibly the largest source of error; however, it is notable that Canadian government agencies used the same values used in the study from Hofmann and Beaulieu (2006).

Table 2-8. Minimum, maximum, and average net livestock inputs calculated from **Error! Reference source not found.**, and the percent deviation from the used value, using the range of values from literature. Original values are from Hofmann and Beaulieu (2006).

	Net P Input per Head					
	Original	Low		High		Average
	(kg P/year)	(kg P/year)	% Change	(kg P/year)	% Change	(kg P/year)
Beef cows	21.9	15.9$	-27.6	21.9*	-	20.0
Dairy cows	27.4	18.4$	-33.1	30.2&	10.0	24.4
Heifers	14.7	4.1#	-72.3	14.7*	-	10.2
Steers	15.7	10.8&	-31.4	15.7*	-	13.3
Bulls	25.8	11.1#	-56.9	25.8*	-	20.9
Calves	7.0	2.4$	-65.5	10.7#	52.3	6.8
Pigs	3.5	1.1&	-69.5	3.5*	-	2.9
Horses	11.8	11.8*	-	11.8*	-	11.8
Sheep	1.5	1.5*	-	1.5*	-	1.5
Chickens	0.1	0.1&	-24.8	0.2*	33.4	0.2
Turkeys	0.6	0.1&	-84.4	0.9$	48.9	0.6

**(Hofmann and Beaulieu, 2006); &(Van Horn, 1998); $(Kellogg et al., 2000); #(Lander et al., 1998)*

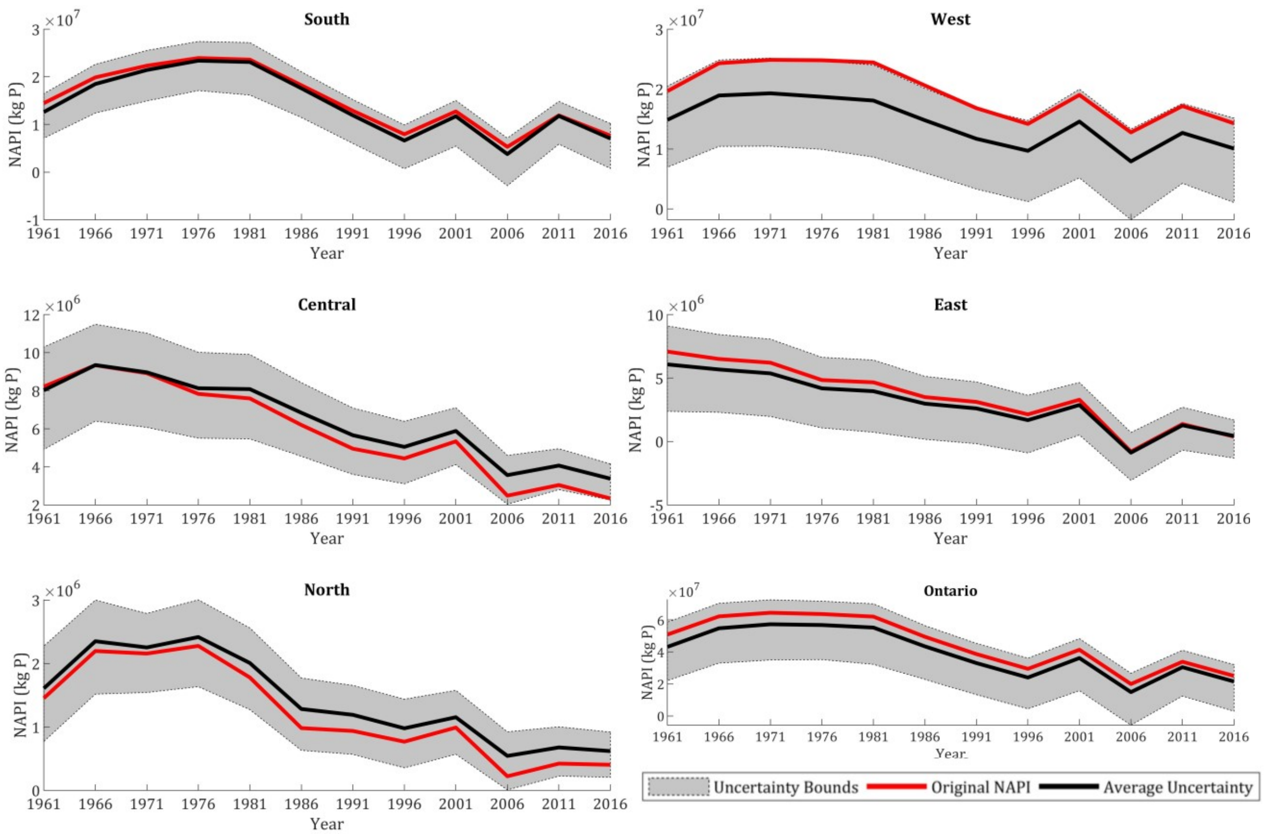


Figure 2-23. Potential deviation from NAPI considering the uncertainties associated with crop yields, regional fertilizer distribution, and net livestock inputs. Livestock inputs include an upper and lower boundary from other potential livestock values, as well as an average of all reported values. Note that each figure has different vertical scales.

2.4 Conclusions

The large-scale spatial analysis fills an information gap about P inputs in Ontario watersheds and their potential to co-determine legacy P accumulation. The data inform decision makers about where legacy P stores might be concentrated. The cumulative map delineates the location of significant P surpluses and potentially at-risk watersheds. The maps can be applied to target nutrient use efficiency, e.g. by aiding producers with their yield optimization costs, and prioritizing soil testing efforts outlined by the 4R methodology. The mass balance results inform government policies about P loading targets for the LGL because they can be used in mechanistic models that estimate long term P loading into the Great Lakes, which are a major binational concern, particularly for Lake Erie. The map time series fill the knowledge gap regarding potential legacy P location and emphasizes the importance of continuous efforts to implement CPs.

Efforts to reduce P from detergents in the 2010 Environment Canada legislation had a noticeable impact on urban inputs in 2011 and 2016. Further research should be conducted to observe the impact of reduced P loading on wastewater treatment plant influent, potentially reducing the pressure on treatment processes that remove P from wastewater, especially in dense population areas such as the Greater Toronto Area.

Increased crop yields have been critical in reducing NAPI in more recent years. However, crop yields are directly influenced by climactic conditions, which are changing with ongoing climate change. In southern Ontario, climate change will likely bring long dry summers with less frequent and larger precipitation events (Shifflett et al., 2012; Water Management Plan Project Team, 2014), which are poor conditions for the crops grown in this region (OMAFRA, 2001). Therefore, the risks associated with climate affecting crop yields in the future could exacerbate NAPI and lead to larger net P inputs.

Fertilizer sales successfully decreased since the 1980s, with 30% less fertilizer used overall. However, an increasing trend in fertilizer use is noticeable in recent years, which, if continued, may offset the progress made unless significant improvements to application practices are made (e.g. applying directly at crop roots). Notably, nutrients from manure decreased over time in many counties due to reductions in cattle numbers, and the many nutrient management plans in place could be a driver for reductions in P losses to surface water. However, the issue remains that ILOs create a discrepancy in the distribution of P by concentrating manure in certain areas, as it is difficult to transport to fields further away that still require fertilization.

A regional analysis of NAPI trends reveals that southern and western Ontario have more agricultural P inputs per unit agricultural area than other regions, influenced by ILOs and a higher proportion of fertilized area. Western Ontario has the highest livestock density, causing it to the only region remaining that did not independently produce enough P to support its human and livestock populations, classifying it as a P importer. Central Ontario remained a P importer as well, although this was driven by the dense population of Toronto.

Analysis of the trends of agricultural and urban practices and land use in the context of legacy P accounts for the consequences of historical practices and informs our understanding of P cycling in urban and agricultural watersheds. Data produced as a result of this analysis will aid in the development of watershed scale legacy P models being developed at the University of Waterloo.

Chapter 3 Accumulation and Loss Risks: Estimating the Fate of Phosphorus in Southern Ontario Watersheds

3.1 Introduction

Legacy P is prolific throughout dense agricultural areas as a result of historical nutrient application. These legacies impact streams, reservoirs, and lakes by triggering HABs through continuous export of nutrients. In particular, P is limiting in freshwater systems (Smith and Schindler, 2009) and has been the focus of many HAB mitigation efforts and regulations. Around the LGL, nutrient reduction targets have been set by Canada and the United States to reduce 2008 loading levels into Lake Erie by 40% by 2025 (Government of Canada, 2018; National Research Council of the United States and The Royal Society of Canada, 2012). An integral part of achieving these targets involves addressing and locating the nutrient legacies that have developed in agricultural soils.

In this chapter, the potential risk of legacy P development in agricultural landscapes was explored using the agricultural elements of the P mass balance developed in the previous chapter. Identifying the intrinsic vulnerability of the landscape on a spatial scale was the first step in estimating risk, inspired by the methods in a similar qualitative analysis (Van Staden et al., 2019). The generated risk maps were used to observe possible legacy P behaviour in the dense agricultural areas of Ontario, as well as two major southwestern Ontario watersheds.

3.1.1 Ontario P mass balance

Delineating the locations of P legacies is important for the purposes of being proactive by employing different CPs to mitigate their negative effects. Identifying and targeting P legacies can be achieved by using a number of methodologies including mechanistic models and high-level multi-criteria risk assessments (MCRA). The first step was to determine P surplus locations and magnitudes, which were delineated for the Province of Ontario with the intention of observing where P was applied in excess within the LGL drainage area (see Chapter 2). It was determined that a significant portion of P surpluses were in the southwestern agricultural regions of Ontario as a result of intensive agriculture, including intensive livestock operations in western Ontario. However, the fate and transport of the P surpluses was not determined and could not be using the mass balance alone.

3.1.2 Vulnerability Models

One approach to determining the fate of legacy P is conducting a risk analysis based on the vulnerabilities of the environmental conditions of the landscape. There are many different methodologies that may be used to conduct a vulnerability and risk analysis. Firstly, a multi-criteria risk analysis (MCRA) considers multiple parameters that would influence vulnerability and combines them together. For example, precipitation, proximity to surface water, slope, soil drainage, erosion potential of soil, proximity to wetland, and presence of tile drains would influence the intrinsic vulnerability of P losses from land surface. These parameters would be reclassified from their quantitative values or qualitative categories (e.g. mm of precipitation, clay soil texture) into new values that represent their relative influence on vulnerability, known as vulnerability indexes (VI). Each parameter may be given more weight using a multiplier if that parameter has a more significant influence on vulnerability than others, otherwise, all parameters should be normalized to have the same weight.

In this case, an alternative to the MCRA is the Revised Universal Soil Loss Equation (RUSLE), which is a model that estimates the probability of soil erosion in different landscapes. Parameters included in the RUSLE are rainfall factor (R), erodibility of soil (K), slope length (L), slope steepness (S), cropping systems (C), and land management (P*). Values for the R, K, and LS factors are calculated, and the C and P* factor values are assigned and combined to compute a final soil loss of erosion potential, similar to the MCRA. The factors are multiplied together instead of summed, and the final value is classified into categories of erosion potential (very low, low, moderate, high, and very high). Precipitation, erodibility, and slope all influence P fate and transport, and overall erosion potential can represent vulnerability to surface losses of P. Generally, the RUSLE was not intended to be used as a vulnerability map, but the development of a spatial representation of the RUSLE was a useful tool that could be used in a vulnerability context.

The intention of the vulnerability and risk analysis was to get a sense of locations with high risk of P loss or P accumulation. The first step was to develop two intrinsic vulnerability maps: one that identified areas in which P would be more likely to erode with soil, and another in which P could accumulate if applied to the land surface. The intrinsic vulnerability maps were then used to create risk maps by combining long term soil P mass balances with the identified intrinsic vulnerability. The results were not intended to provide numerical values of P runoff and erosion losses per hectare. Instead, it suggested areas that would have a higher risk of P loss or accumulation based on a relative risk rating scale represented by the intrinsic vulnerability of the land surface and the presence of the P contaminant.

3.1.3 GRW and TRW

The Grand River Watershed (GRW; 6,800 km^2) and the Thames River Watershed (TRW; 3,089 km^2) are the two largest Canadian watersheds that drain into Lake Erie. The GRW and the TRW are both heavily studied watersheds in southwestern Ontario because they are significant contributors of P to Lake Erie. The GRW is the largest Canadian watershed that drains into Lake Erie and has been commonly held as the largest contributor of P to the eastern basin (Loomer and Cooke, 2011). The TRW is a large contributor of P to Lake St. Claire, which drains into Lake Erie. It should be noted that P loadings from the American side make up approximately 80% of the loads to Lake Erie (Maccoux et al., 2016), however, the contributions from the Canadian side should not be discounted because of this.

Legacy nutrients are prevalent in both the GRW and TRW, offsetting the CP efforts to reduce P loading. The time difference between nutrient reduction efforts on land and the time the nutrient loading reductions are seen at the stream outlet is referred to as a lag time (Van Meter and Basu, 2017). Lag times are driven by the influences on the fate and transport of nutrients; they are increased by slow transport times in groundwater and other retention mechanisms, and decreased by the presence of tile drains (Van Meter and Basu, 2017). One previous study done in the GRW identified lag times ranging from 10 to 34 years, with higher lag times in the upper and lower areas of the catchment (Van Meter and Basu, 2017). Similar lag times are plausible in the TRW, which has similar land use and tile drains present.

3.2 Methods

3.2.1 Site Description

The Province of Ontario has a significant proportion of the Canadian population living in dense urban centers close to the LGL. Agriculture also concentrates in the same centers, which has led to significant P contributions to Lake Erie over time, and consequently, high risk HAB conditions. The landscape characteristics of Ontario influence the fate and transport of P, including topography, climate, and surficial geology.

The elevation of Ontario ranges from sea level to just under 700 m in the northern region at Maple Mountain in Timiskaming District (Baldwin et al., 2000). Blue Mountain in the western region of Ontario notably has a much higher elevation than the surrounding area, and steeper slopes moving outward, whereas the slopes closer to the LGL flatten out significantly (Baldwin et al., 2000). Northern Ontario has the lowest elevation along the edge of the Hudson Bay; however, the populated areas in northern Ontario have high elevations with steep slopes. The lowest elevation outside of northern Ontario is the eastern Ontario region near Ottawa, in which the slopes are gradual as well.

Generally, the climate in Ontario is humid continental (Baldwin et al., 2000), with the highest temperatures in the southwestern-most tip of the province in Essex county (Baldwin et al., 2000). The mean daily temperature in Ontario follows a decreasing temperature gradient northward, with similar temperatures in the Sudbury and Thunder Bay areas closest to the Great Lakes. Precipitation follows a similar trend northward, ranging from 500 to over 1000 mm annually (Baldwin et al., 2000). The largest amounts of annual precipitation fall closer to the Great Lakes, particularly the eastern edges of Georgian Bay and Lake Huron, as well as the eastern edge of Lake Superior near Sault Ste. Marie (Baldwin et al., 2000). However, summer precipitation trends are very different, with more precipitation falling along the eastern side of northern Ontario, away from the Great Lakes.

The surficial geology is diverse throughout Ontario: from the Canadian shield with the oldest exposed rock in Canada to the fine glacial sediment that surrounds the LGL (Figure 3-1). Northern Ontario is dominated by bedrock, with spots of coarse sediment and till. In southern Ontario, where agriculture is more concentrated, the sediment is comprised of fine-grained sediment, as well as some coarser sediment and till (Neff et al., 2005). Around the edges of Lake Erie, the surficial geology is dominated by lacustrine deposits and other fluvial deposits as a result of glacial activity (Baldwin et al., 2000). Ground moraine, a till comprised of clay and sand, is found across southern Ontario extending to the southwestern tip (Baldwin et al., 2000). In western Ontario, with the higher elevations, the surficial geology is dominated by end moraine, or coarser debris deposited at the outer extent of the glacier (Baldwin et al., 2000).

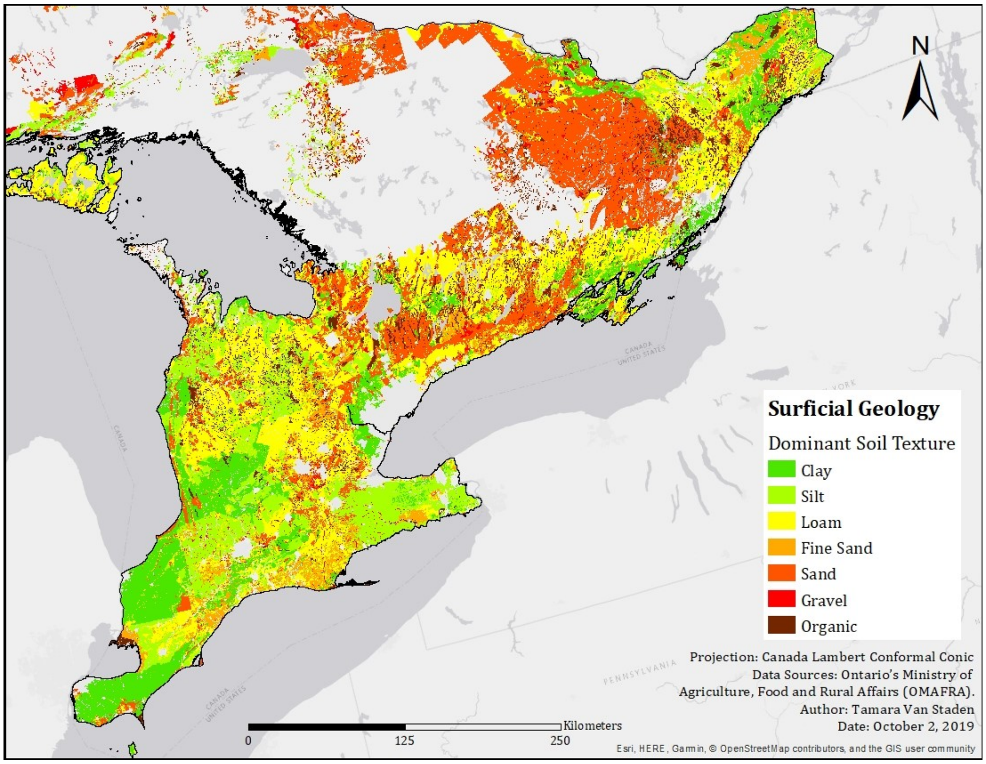


Figure 3-1. Surface soil texture in Ontario. Data from OMAFRA (Saurette, 2015).

3.2.2 Vulnerability Map Development

3.2.3 Multi-Criteria Risk Assessment (MCRA) – The Swedish Example

As a pilot study for developing a vulnerability map in the context of contaminant losses from land instead of aquifer vulnerability, a MCRA was conducted on the Norrström basin in Sweden. An initial intrinsic vulnerability map was developed using the MCRA to indicate vulnerable areas. The vulnerability map could then be compared to a cumulative P surplus map to generate both P erosion and accumulation risk maps. Due to time constraints, the Norrström basin vulnerability map was not compared to a P surplus map in this study, but research for this project is ongoing.

As mentioned, the developed model considered precipitation, proximity to surface water, vulnerability to soil erosion, slope, soil drainage, proximity to wetland, and presence of tile drains to evaluate the intrinsic vulnerability of P losses from land surface. Combined, the layers represent the intrinsic vulnerability of P losses in the Norrström basin. The layers for proximity to surface water (P2SW), precipitation (P_v), slope and drainage (SD), erosion vulnerability (EV), and tile drain (TD) presence were additive, and result was multiplied by the wetland drainage layer (WD):

$$Intrinsic\ Vulnerability = (P2SW + P_v + SD + EV + TD) \times WD \qquad \text{Equation 11}$$

The P2SW, P_v, SD, EV, and TD parameters were considered additive because they affect surface losses and increased P loss vulnerability, while the WD parameter decreased vulnerability as wetlands act as an ecosystem service that increase P storage capacity. The wetland parameter was multiplicative because the value assigned reflected the percent overall reduction in P losses.

Spatial data of each parameter were mapped and simplified into numeric categories, referred to as reclassification, which represented effects on the fate and transport of P. All parameters were represented as rasters in GIS and the values were summed together to create a map of relative vulnerability, as described in (Focazio et al., 1984). All layers were clipped to the basin in the projection Lambert Azimuthal Equal Area and areas covered by lakes were removed from the dataset.

3.2.3.1 The Norrström Basin

The Norrström basin in Sweden is west of Stockholm and drains into the Swedish archipelago. The landscape around the basin includes mountain ranges to the northwest, the Baltic Sea to the east, and large lakes in the southern side of the basin. Agriculture is prevalent around the lakes further away from the mountains. The terrain is rocky in parts and dominated by glacial sediments.

The Norrström basin is dense in agriculture and is therefore exposed to the risks of excess nutrient deposit from non-point sources. Eutrophication is a major concern for the Baltic Sea, similar to Lake Erie, and conducting a similar mass balance and vulnerability analysis may contribute to mitigation efforts for the region.

3.2.3.2 Reclassification

Each layer was reclassified in order to represent the influence of each parameter under the intrinsic vulnerability analysis. Each spatial layer was reclassified from their original value to one which reflects the potential impact it would have on P mobilization, with higher values indicating higher vulnerability. Reclassification values, or VIs, were set for each parameter. Each layer was rasterized (spatially discretized) and each raster cell was reclassified using the values set (see Table 3-1 – Table 3-6). Reclassifying and combining the layers allows for a relative representation of vulnerability that has the potential to indicate areas of concern (Focazio et al., 1984).

3.2.3.3 Precipitation (P_v)

Daily precipitation data were available from weather stations across Sweden through the Swedish Meteorological and Hydrological Institute (SMHI). For the Norrström basin, 178 weather stations were selected within the basin itself and within a 50 km buffer of it. Data were selected for the most recent available year, 2010, for two separate time periods: the cold season (January 1 – March 31 and October 1 – December 31) and the growing season (April 1 – September 30). These dates, defined by Alberta Agriculture and Forestry, were used assuming similar climate in both regions.

To determine the impact of precipitation, a parameter known as the "precipitation variable" (P_v) was used, developed by Van Staden et al. (2019). The P_v was calculated by multiplying the average daily precipitation by the variance in daily precipitation, excluding days with no precipitation. One P_v value was calculated for each season at each weather station. In areas where a high average daily precipitation and a high variance coincided, it was likely that very large precipitation events occurred in that area over the season. Conversely, a low average and a low variance would indicate there were smaller events in that area. Larger event sizes would increase the risk of P runoff; therefore, high values were assigned a high vulnerability.

The vulnerability index for precipitation was determined using the histogram distribution of precipitation and influence from similar studies (Ebert et al., 2016; Van Staden et al., 2019). In previous works, a P_v greater than 400 mm^3 was given the maximum VI; there were a few large precipitation events in the Norrström basin dataset, with some P_v values exceeding 2,500 mm^3. Using natural breaks in the dataset combined with influence from previous studies, VI category values of P_v were selected (Table 3-1).

The weather station point data were interpolated by kriging after the P_v was calculated for each station. Semi-variograms were constructed for both seasons and it was determined that the cold season did not have a good fit for interpolation. Additionally, as P should not be applied to land when the ground surface is frozen, the growing season precipitation was determined to be the most appropriate dataset to use. Therefore, only data from the growing season (April 1 – September 30) were considered in the model.

Table 3-1. Vulnerability indexes for the precipitation variable set as a pseudo-log scale with increasing vulnerability for larger values.

Range (mm^3)	Vulnerability Index
0 – 1	0
1 – 10	1
10 – 100	2
100 – 200	3
200 – 400	4
>400	5

3.2.3.4 Proximity to surface water (P2SW)

The distance that P must travel from its application on agricultural fields to surface water directly influences the likelihood of P reaching the surface water. Distance to surface water was estimated using the Hydrology toolbox in ArcGIS, and the value in each raster cell represents the total distance along the downslope flow path from the cell location to its outlet at the closest surface water body. First, a DEM of the study area was processed through the Sink and Fill tools to eliminate imperfections in the DEM. The Flow Direction tool was used to generate a layer that indicated the direction of water flow from each raster cell. The flow direction layer was then used to estimate the total distance to surface water in the Flow Distance tool. The result was a raster with 30m resolution indicating the approximate downslope distance to the nearest lake or river.

The two highest vulnerability indexes were set using recommendations from OMAFRA regarding appropriate P application setback distances from surface water under different soil P conditions (Hilborn and Stone, 2005). Under conditions where P is likely to move from the application site, it is recommended that P is applied at a 30 m setback if meeting crop needs and a 60 m setback if the P application would exceed crop needs (Hilborn and Stone, 2005). The remaining distance values were set based on reasonable distances that surface water could travel as overland flow during a precipitation event (Table 3-2).

Table 3-2. Vulnerability indexes for setback distances from surface water.

Range	Vulnerability Index
0 – 30 m	4
30 m – 60 m	3
60 m – 1 km	2
1 km – 5 km	1
>5 km	0

3.2.3.5 Slope and Drainage (SD)

The slope of the landscape affects the likelihood of applied P moving in overland flow during a precipitation event. However, P can also seep into the subsurface, therefore, the drainage of the land surface was considered in the model. The vulnerability indexes were assigned using the water runoff classes from OMAFRA (Table 3-3) (Hilborn and Stone, 2005).

Table 3-3. Vulnerability indexes for hillslopes under different surface drainage classes, values from (Hilborn and Stone, 2005).

Drainage	**Rapid (A)**	**Moderate (B)**	**Slow (C)**	**Very Slow (D)**
Slope	Vulnerability Index			
< 3%	1	1	2	4
3 – 6%	1	2	4	8
6 – 9%	2	4	8	8
> 9%	8	8	16	16

The adjusted DEM was used to determine the slopes within the Norrström basin using the Slope tool in the Hydrology toolbox in ArcGIS. The drainage classes of the land surface were determined using surficial geology data from the Geological Survey of Sweden (SGU) (Table 3-4). The land surface classifications were translated and assigned an appropriate drainage class to be used for the vulnerability index reclassification (Table 3-3).

Table 3-4. Surface geology categories from the SGU and the assigned drainage class.

SGU Categories	**Translation**	**Drainage Class**
Isålvssediment	Ice sediment	Moderate (B)
Torv	Peat	Rapid (A)
Postglacial sand-grus	Glacial sand-gravel	Rapid (A)
Lera-silt	Clay-silt	Very Slow (D)
Vatten	Water	NA
Morän	Till	Moderate (B)
Berg	Rock/mountain	Very Slow (D)
Moräniera eller lerig Morän	Muddy till	Slow (C)

3.2.3.6 Clay and Erosion Vulnerability (EV)

Fate and transportation of P to surface waters is dependent on soil characteristics such as clay content (Vickers, 2017). The tendency of P to attach to clay in soil reduces its mobility and would therefore reduce the vulnerability of P losses to surface water. At first, it was assumed that more clay linearly decreases the vulnerability of P losses. The VI was assigned by subtracting percentage of clay in the soil from the total (100%). For example, if there is 10% clay in the soil, the vulnerability index is 0.9.

Notably, this clay layer would affect dissolved P the most, whereas the model considers total P, so an alternative method was selected, considering erodibility using Hjulström's diagram (Hjulström, 1935). The diagram states that the particle size at the lowest water velocity where erosion occurs is 0.1mm or fine sand. Larger particles require a higher water velocity to erode because of increasing particle weight towards a larger grain size, and smaller particles require a higher water velocity due to increasing cohesive forces towards a smaller grain size (i.e. clay and silt). Vulnerability to erosion would be at a maximum at 0.1 mm and would decrease towards both decreasing and increasing particle size.

Detailed soil textures from the SGU were classified from their descriptions to generalized categories and assigned a vulnerability index based on Hjulström's diagram (Hjulström, 1935) (Table 3-5). For instance, where the SGU soil descriptions did not specify sand grain size, a spatial analysis was conducted to observe the characteristics of nearby soil samples. Assuming that "near things are more related than distant things" (Tobler's First Law of Geography), the grain size characteristics of nearby polygons were applied to the polygon in question. The final vulnerability index values assigned to the unique descriptions in the detailed SGU soil texture shapefile can be found in Appendix B (Table B-1).

Table 3-5. Vulnerability indexes for erosion potential of different soil textures based on Hjulström's diagram (Hjulström, 1935).

Soil type	Vulnerability Index
Clay	1
Clay-Silt	1.5
Silt	2
Fine sand (0.1 mm)	3
Medium sand	2
Coarse sand	1
Gravel	0
Exposed bedrock	0

3.2.3.7 Wetland Drainage (WD)

Wetlands provide an ecosystem service by retaining P through biological interactions with wetland vegetation and chemical interactions with wetland soil, among many mechanisms (Reddy et al., 1999). The function of retaining P has had a significantly positive impact on downstream surface water by slowing P transport and cycling it within smaller ecological systems in the watershed (Reddy et al., 1999).

One Ontario study found that a net 25% of P loadings were retained in the wetland (Gehrels and Mulamoottil, 1990). Using this information, wetland drainage areas within the Norrström basin were classified to reduce the intrinsic vulnerability. Areas within 5 km of the drainage system of a wetland had vulnerability reduced by a maximum of 25% to account for the total P retention capacity in those areas. The proximity to surface water methodology was used to scale the vulnerability reduction multiplier for distances within 30 m, 60 m, 1 km, and 5 km of the wetland (Table 3-6). The total flow distance along the flow path towards the wetland was estimated, and areas that were further than 5 km away were removed from the layer.

The wetlands layer did not contribute to the vulnerability, but instead functioned as a parameter that reduced the total risk of P reaching other surface waters, and therefore was considered a multiplier to the vulnerability analysis. The multipliers were set to remove 25% from total vulnerability, ranging from a multiplier of 0.75 closest to the wetland for maximum reduction, to a multiplier of 1 more than 5 km from the wetland to keep the vulnerability the same. The total intrinsic vulnerability value was multiplied by the wetland layer at the final step to reflect the final intrinsic vulnerability.

Wetlands could alternatively be considered as surface water in the proximity to surface water layer if the wetlands are considered at risk. However, in this study, wetlands were considered solely to provide the ecosystem service of P storage to the basin. As mentioned, it should be noted that the benefits of wetland P retention are limited, and they could eventually become sources of P.

Table 3-6. Vulnerability reduction multiplier values for setback distances from wetlands.

Range	Vulnerability Multiplier
0 – 30 m	0.75
30 m – 60 m	0.8
60 m – 1 km	0.85
1 km – 5 km	0.9
>5 km	1

3.2.3.8 Tile Drainage (TD)

Tile drains are constructed in agricultural lands where the soil texture is poorly drained to improve the productivity of the land (Sugg, 2007). The channels allow for water to move easily through the subsurface so that the water table does not reach the root zone in agricultural soil, which would lead to anoxia (Agriculture and Agri-Food Canada, 2009). A consequence of drained soil is that it allows particles to move more easily from the surface, including nutrients. One Canadian P transport model assumed that the presence of tile drains provided a direct route for nutrients to surface water (Agriculture and Agri-Food Canada, 2009). In this model, the presence of tile drains was assumed to increase the vulnerability of P losses.

Spatial tile drain data are not available for the Norrström basin, so an estimation of tile drain locations was conducted based on drainage properties of the soil in agricultural land use area, and county-reported tile drain coverage. Following the methods of Sugg (2007), it was assumed that the soil textures with the poorest drainage in agricultural land use areas would be the best candidates for tile drainage locations. Tile drainage survey data from 2013 was available for the counties in the Norrström basin. Total land area of poorly drained soils (i.e. clay, silt, and peat) was compared with reported land area of tile drains from the survey, and the probability for tile drainage presence was assigned to the soil textures, with the highest probability assigned to clay, followed by peat and silt. Soil textures from the SGU were used to correlate with potential tile drain systems and the probability was estimated within cropland areas reported in 2018-2019 remote sensing surveys. The probability values ranged from 0 to 1 and were directly assigned as the VI for each location. It was assumed that if a tile drain was present, it would effectively set the proximity to surface water to zero, and therefore would have maximum vulnerability.

3.2.4 RUSLE – Erosive Potential as Vulnerability

Potential soil erosion maps were developed by OMAFRA using the RUSLE methodology. The maps were created by multiplying the R, K and LS factors estimated for the province. It is important to note that the cropping systems (C) and support practices (P*) factors were not estimated for Ontario at this time due to the complex spatial distribution of C and P* in the province, and the temporal nature of the data required for province-wide estimates of these factors. For this reason, the potential soil erosion reported was the maximum potential soil loss, given that the C and P* factors are used to reduce potential soil loss based on crop cover and soil management. For example, agricultural areas in silage corn with conventional tillage operations would see the erosion estimate be multiplied by a factor of 0.56 (Wall et al., 2002), thus resulting in a reduction of the estimated potential soil loss.

3.2.4.1 *Risk Map Development*

Risk considers both the vulnerability of the land, erosion in this case, and the magnitude of the threat, P surpluses. This can be calculated as "Risk = Vulnerability x Threat". The intrinsic vulnerability map was combined with the P mass balance by multiplying the corresponding values in each raster cell. Before multiplying, the RUSLE map layer was reclassified from numeric values of erosion (in tonnes/hectare/year) to soil erosion classes modified from those presented in the RUSLE2 handbook (Wall et al., 2002) (Table 3-7). The classes were based on whether the potential soil erosion amount would have a significant impact on soil productivity and soil loss tolerance (Wall et al., 2002). For this study, the RUSLE vulnerability layer was reclassified into more useful and understandable categories. The class breaks were adjusted slightly from those reported in Wall et al. (2002) in consultation with OMAFRA (McKague, personal communication).

Assigned vulnerability indexes increased with higher erosion potential from 1 to 6 to represent vulnerability to P losses by erosion and decreased with higher erosion potential from 6 to 1 to represent vulnerability to accumulation as a result of little to no erosion losses (Table 3-8). There are limitations in the assumption that a lack of erosion increases the vulnerability to accumulation as there are other subsurface processes and transport mechanisms that influence the fate of the identified P surpluses, and the subsequent P species introduced to the environment. However, this approach was deemed acceptable as a relative representation of intrinsic vulnerability.

Table 3-7. RUSLE potential soil erosion from Wall et al. (2002).

Erosion potential (t/ha/year)	Class
0 – 6	Very Low
6 – 11	Low
11 – 22	Moderate
22 – 33	High
> 33	Severe

Table 3-8. Reclassification of RUSLE erosion potential for both erosion and accumulation vulnerability.

Erosion potential (t/ha/year)	Class	Loss by Erosion	Accumulation
0 – 2	Negligible	1	6
2 – 6	Very Low	2	5
6 – 11	Low	3	4
11 – 22	Moderate	4	3
22 – 33	High	5	2
> 33	Severe	6	1

For this study, the RUSLE vulnerability layer was reclassified into more useful and understandable categories. A similar reclassification was done for the cumulative P mass balance layer. The weight of each layer was made to be the same, so the index assigned to the highest category was set to the same value of 6 (Table 3-9). The categories were based on recommended soil P concentrations in the Agronomy Guide for Ontario (Brown et al., 2017). The maximum soil P concentration on which fertilizer application was recommended in the Agronomy Guide was 60 ppm (approximately 120 kg/ha using the conversion from Oryschak et al. (2008)), much lower than the highest cumulative P value from the mass balance (1053 kg/ha). To address this, it was assumed that 10% of the cumulative P was representative of what remained in the top 15 cm of the soil, and the original categories were included and extended to ten times the P concentration of each category, up to 1200 kg/ha (Table 3-9). The increase in VI from the second last to the last category was larger than between other categories to keep the maximum value of 6, and reasonably captures the significant increase in cumulative P.

Table 3-9. Reclassification of the cumulative P balance based on crop responses to fertilization in different soil P concentrations (Brown et al. 2017). Soil concentration conversion from Oryschak et al. (2008).

Cumulative P (kg P/ha)	**VI**	**Potential P Excess of Crop Needs**	**P in soil if 10% of cumulative retained** (ppm)	**Crop response to fertilizer application** (Brown et al., 2017)
0 - 18	1	Negligible	0.9	High
18 - 30	1.5	Minimal	1.5	High
30 - 50	2	Very low	2.5	High
50 - 80	2.5	Low	4	High
80 - 120	3	Moderate Low	6	High
120 - 180	3.5	Moderate	9	High
180 - 300	4	Moderate High	15	Medium
300 - 500	4.5	High	25	Medium Low
500 - 800	5	Very High	40	Low
800 - 1200	6	Severe	60	None

3.2.5 P Erosion Estimation

In the interest of estimating soil P erosion magnitudes, possible masses were estimating using annual soil test P (STP) data as Olsen P (ppm) from Agriculture and Agri-Food Canada (AAFC) from 1981 to 2011; bulk density values, from the Soil Landscapes of Canada (SLC); and erosion potential, from the RUSLE layer provided by OMAFRA. The STP values were converted to kg P/m^3 of soil using the conversion 1ppm = 0.001 kg/m^3. Then the kg P/m^3 value was divided by the bulk density of the soil (BD) to get kg P/kg soil. Finally, kg P/kg soil was multiplied by the erosion potential (EP) to get kg P eroded per hectare. The potential eroded P mass was calculated for each year and summed to estimate a total mass of eroded P from 1981 to 2011 using the following equation:

Equation 12

$$\sum_{i=1981}^{2011}\left(STP(ppm)_i \times 0.001\left(\frac{kg\ P}{ppm \cdot m^3\ soil}\right) \times \frac{1}{BD}\left(\frac{m^3\ soil}{kg\ soil}\right) \times EP\left(\frac{kg\ soil}{ha}\right)\right) = Eroded\ P\ \left(\frac{kg\ P}{ha}\right)$$

where STP is soil test P, BD is bulk density of soil, and EP is erosion potential.

The cumulative agricultural P surplus from 1981 to 2011 was calculated from the mass balance data and compared to the estimated soil P erosion mass from the same time period. The total P surplus was summed across the province and compared to the cumulative sum of soil eroded P. The difference between agricultural P surplus inputs and estimated soil P erosion over the same time period represented the P that remained in the system, or that had been transported by other means.

3.3 Results and Discussion

3.3.1 Vulnerability maps

3.3.1.1 MCRA

Six parameters were included in the MCRA in the Norrström basin, and each had a unique spatial distribution of vulnerability (Figure 3-2). The proximity to surface water layer had high vulnerability along the rivers, with scarce low vulnerability areas where the flow path from the source to the surface water was long and tortuous (Figure 3-2a). For precipitation, there was a higher vulnerability, and therefore larger precipitation events in the southern area of the basin, and closer to the Baltic Sea to the east (Figure 3-2b). The slope index layer had spots of high vulnerability in the northwest and southeast quadrants, where there were both steep slopes and poorly drained sediments (Figure 3-2c). In particular, clay dominated sediments are prevalent in the northwest quadrant of the Norrström basin (Ballabio et al., 2016), driving this outcome. Erosion vulnerability was highest where surficial geology was dominated by a fine sand texture (Hjulström, 1935), mostly in the southwestern quadrant, and sparse in the clay-dominated northwestern quadrant (Figure 3-2d). Wetland drainage areas within five kilometers took up approximately 37% of the basin, with only 5% of the basin within very close proximity (< 30 m) to a wetland (Figure 3-2e). Areas within a closer proximity to a wetland had a larger reduction in vulnerability. Tile drains location probability was estimated within cropland area and some locations had notably high probability of tile drain presence (Figure 3-2f). Most high values only approached a probability of 1; very few areas were assigned a 100% probability of tile drain presence.

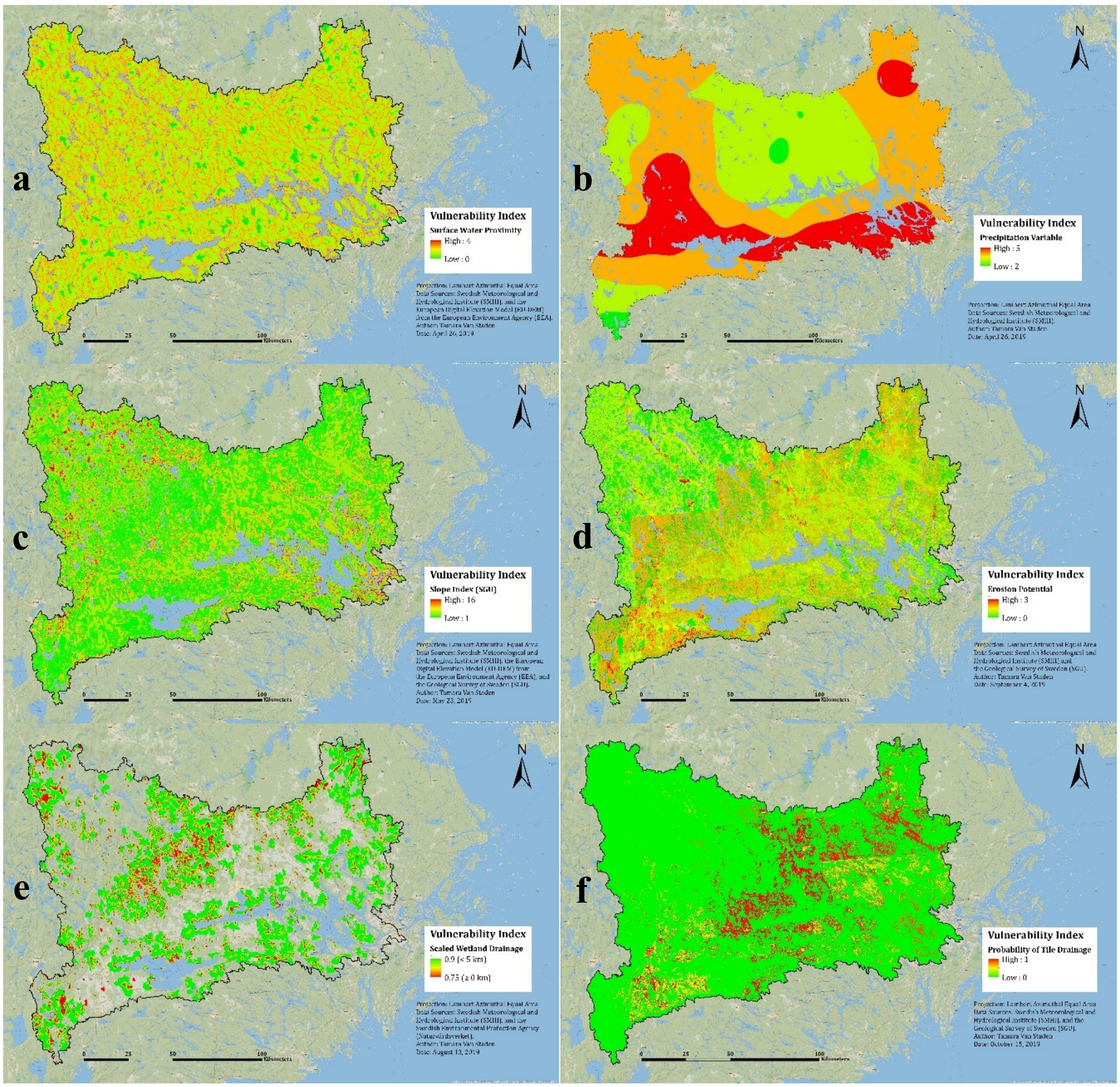


Figure 3-2. Parameters for the multi-criteria risk assessment conducted in the Norrström basin in Sweden: a) Proximity to surface water; b) Precipitation variable; c) Slope and drainage; d) Erosion vulnerability; e) Proximity to wetland; and f) Probability of tile drain presence.

The layers were normalized to values between 0 and 1 to ensure that each layer had equal influence on vulnerability, and then combined as described in Equation 11. The final map showed areas with a higher vulnerability to P losses to surface water, specifically, where tile drain presence, high precipitation, steep slopes, and erosive soil textures overlapped (Figure 3-3). The highest vulnerability was observed in the southwest quadrant in high precipitation areas close to rivers (Figure 3-2a and b). The maximum value that could be calculated using Equation 11 was 5, where all parameters had maximum vulnerability and were not within the drainage system of a wetland. However, the maximum value estimated within the Norrström basin was 4.7, meaning that the maximum potential vulnerability was not reached for this area.

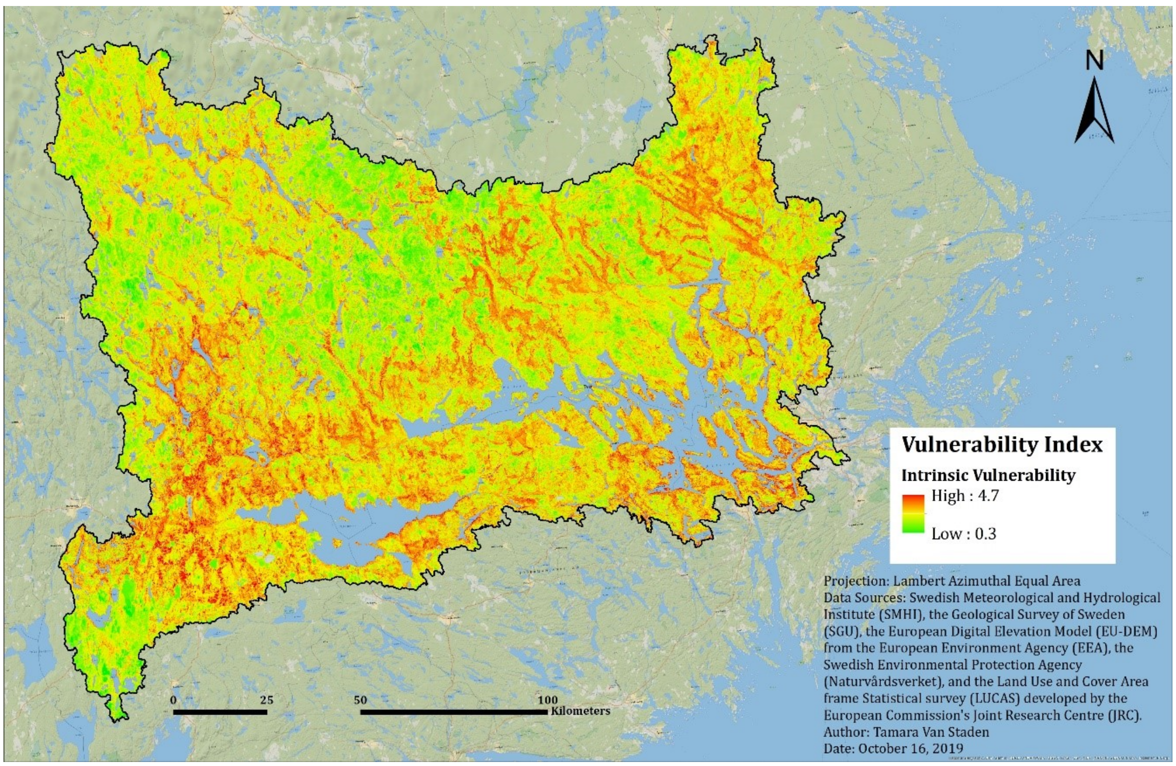


Figure 3-3. Multi-criteria intrinsic vulnerability map of potential P losses from land surface to surface water in the Norrström basin, Sweden.

3.3.1.2 RUSLE

The potential soil erosion values as tonnes/ha/year were classified into qualitative categories that reflected the relative impact of erosion on the soil (Table 3-8). The developed maps reflected the vulnerability for P to be lost by erosion (Figure 3-4) and the vulnerability of soil to accumulate P due to lack of erosion (Figure 3-5).

The risk of erosion decreased southward towards Lake Erie and eastward from Toronto (Figure 3-4). The decrease in erosion risk from western to southern Ontario was driven by lower slopes and a decrease in grain size as clay dominated soils become more prevalent (Baldwin et al., 2000). Erosion potential was low closer to the Lake Erie edges (Figure 3-4) where there are more lacustrine sediments and clay (Baldwin et al., 2000). Western Ontario closest to Lake Huron and the Georgian Bay had the highest risk of erosion as a result of larger grain size and steeper slopes (Baldwin et al., 2000). In eastern Ontario, the erosion risk was mostly negligible as a result of the elevation and slopes both being gradual (Figure 3-5).

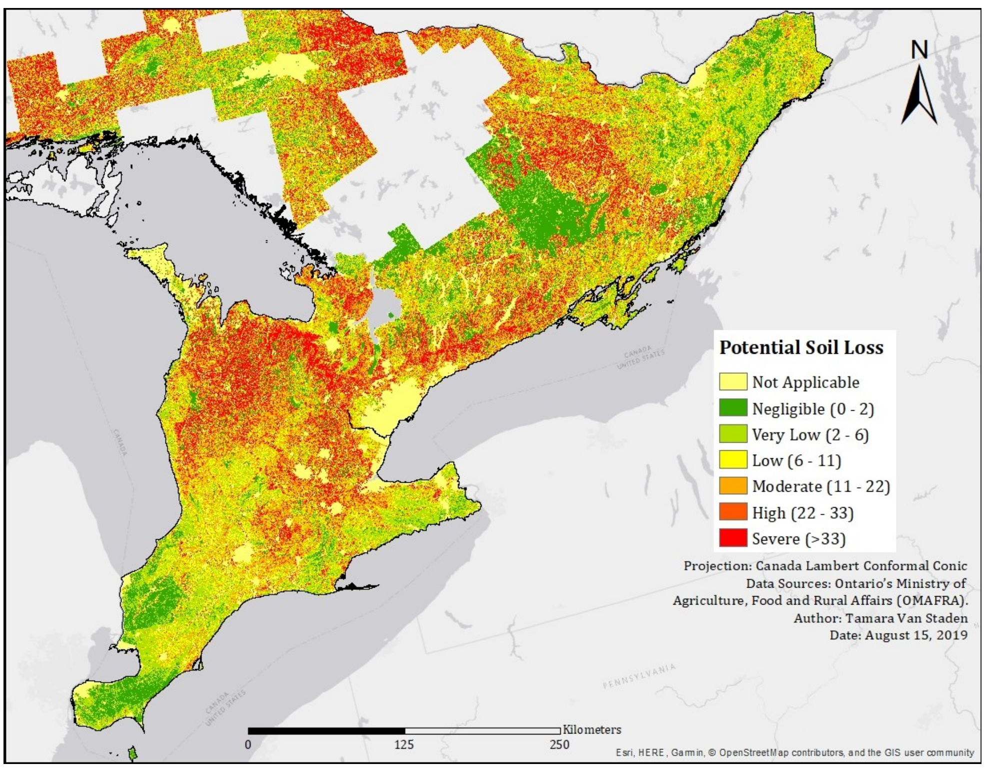


Figure 3-4. Map of the potential soil loss in agricultural areas of Ontario based on RUSLE. Data provided by OMAFRA, Government of Ontario (2019).

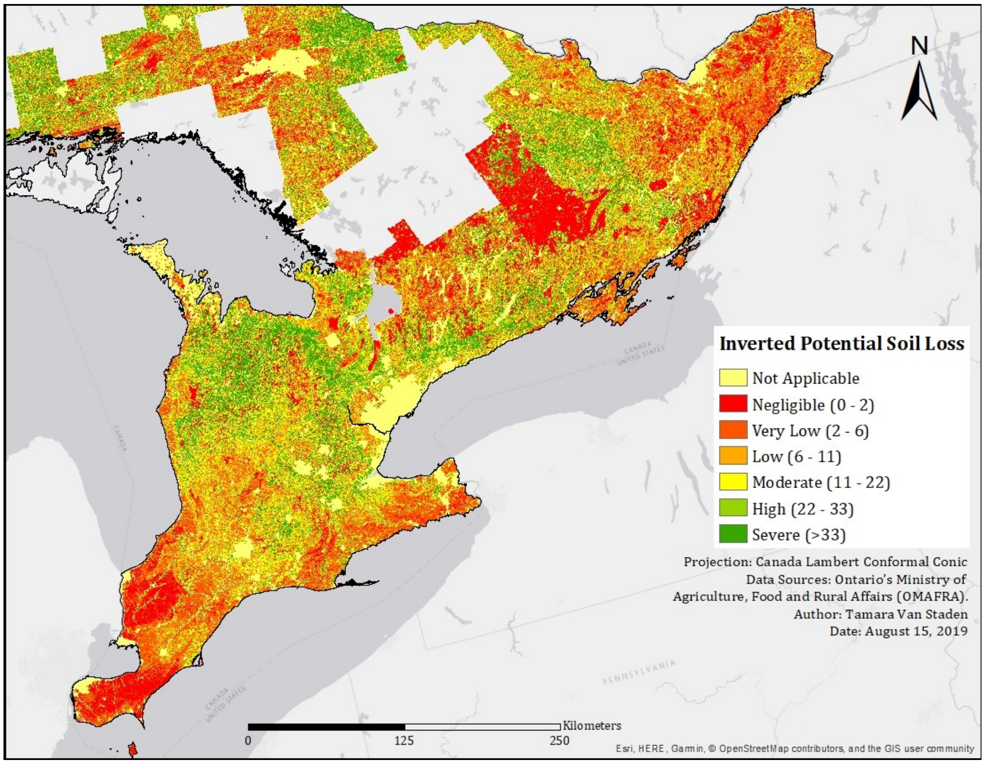


Figure 3-5. Map of the inverted colour scheme of potential soil loss, representing potential accumulation of P in soils. Data provided by OMAFRA, Government of Ontario (2019).

3.3.2 P Surplus Threat Map

The previously estimated cumulative P surplus represents the "Threat" component for the calculation of risk: "Risk = Vulnerability x Threat". The presence of P as contaminant was critical in order to observe the influence of the intrinsic vulnerability on the fate of P.

The cumulative P surplus reclassification isolated the three counties with the largest surplus: Waterloo Region, Perth County, and Oxford County. The distribution of the classes decreased outwards from the high surplus hub in southwestern Ontario, reaching its lowest point in Prince Edward county, and increasing again in eastern Ontario (Figure 3-6). The focal point of high P surpluses in southwestern Ontario existed as a result of disproportionate P application in the agricultural region. More manure P was produced there than was able to be used efficiently, and regions further away were not able to import the manure and needed to purchase nutrients as synthetic fertilizer.

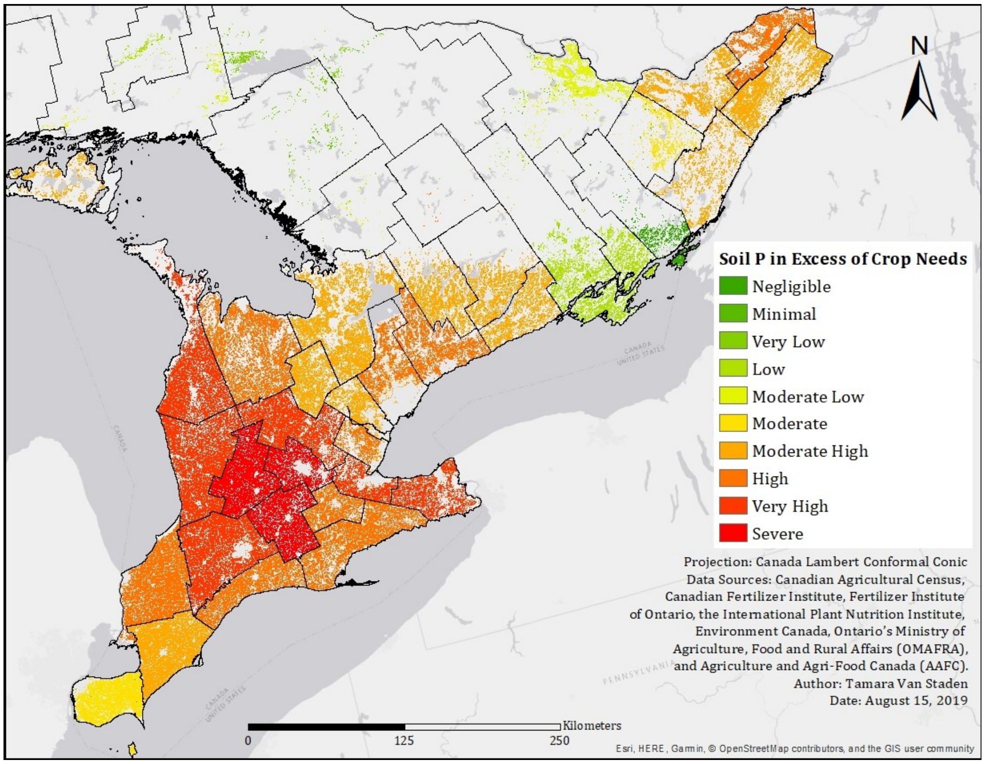


Figure 3-6. Map of reclassified agricultural cumulative P balance in the southern Ontario regions. The classifications were based on recommendations from the Agronomy Guide for Ontario (Brown et al., 2017).

3.3.3 Risk Maps

The risk values ranged from 0 to 36 in both risk maps and a clear inversion was visible between the risk of erosion map (Figure 3-7) and the risk of accumulation map (Figure 3-8). Generally, risk of erosion concentrated in western Ontario, and risk of accumulation concentrated in southern and eastern Ontario. Note that these maps primarily target erodible P, primarily in particulate form.

The lacustrine sediments along the Canadian side of Lake Erie were compressed by the Laurentide glacier, and the surficial geology of these areas is now dominated by clay and sand plains (Stone and Saunderson, 1992). The fine grain size combined with shallow slopes resulted in higher accumulation risk in these areas, despite lower cumulative P surplus. In addition, the presence of clay would affect dissolved P, causing it to be retained in in the soil (Reid et al., 2018; Vickers, 2017). The areas with P accumulation have a higher probability of being the same areas with the legacy P that has affected Lake Erie.

Erosion was higher near the eastern edge of Lake Huron and the southern edge of the Georgian Bay (Figure 3-4) where there is more relief and slopes tend to be longer and fairly steep. The surficial geology in this area is dominated by coarser moraines and limestone plains that are more subject to erosion (Baldwin et al., 2000; Stone and Saunderson, 1992). These areas in conjunction with excessive P surpluses (Figure 3-6) had high erosion risk and therefore were at greater risk of losing any P that had been applied there or had accumulated in soils previously.

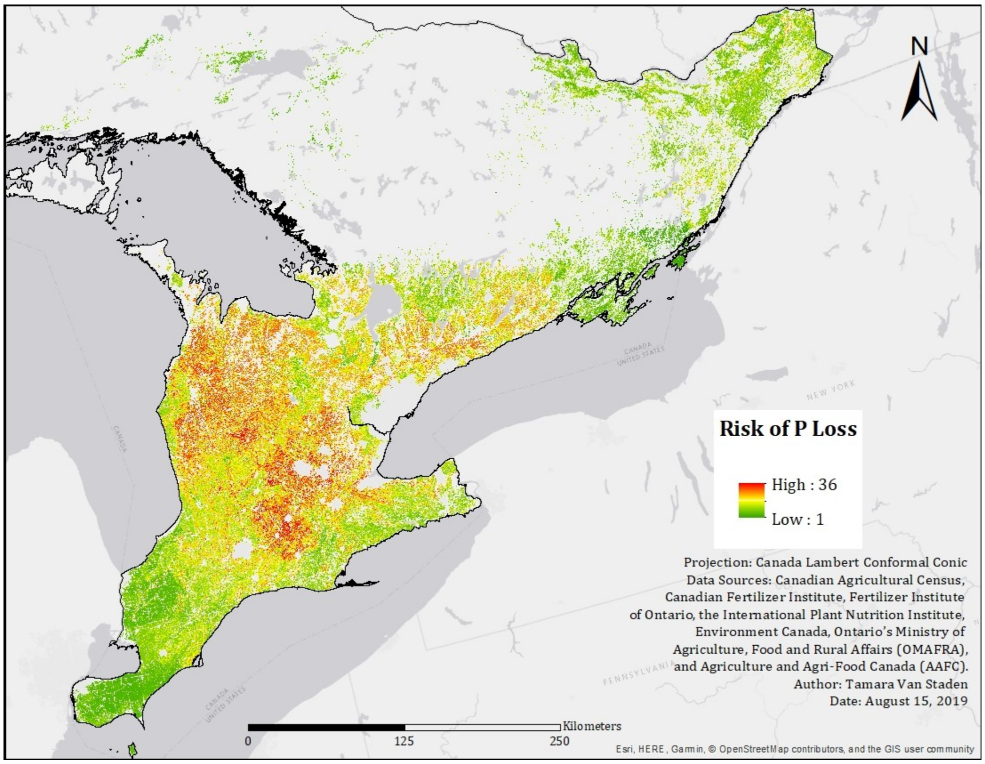


Figure 3-7. Risk map of P losses due to erosion in the southern Ontario agricultural regions.

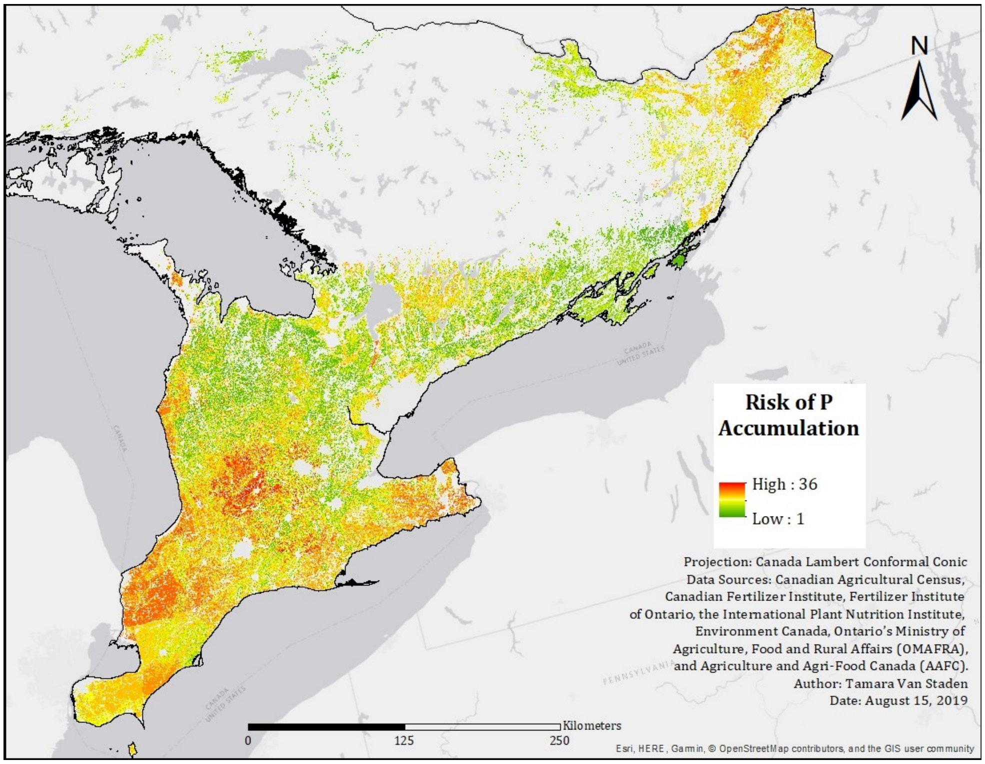


Figure 3-8. Risk map of P accumulation due to lack of erosion in the southern Ontario agricultural regions.

3.3.4 GRW and TRW Trends

The watershed area of both the Thames River and the Grand River were mostly covered by either type of the highest risk category. Both watersheds are important aquatic ecosystems in southern Ontario that are highly monitored by conservation authorities.

The GRW had both high erosive risk and high accumulation risk with the former covering more area (Figure 3-9). Due to there being more erosive risk in the GRW, there was a higher probability that P losses in the GRW occurred as a result of soil erosion rather than losses as dissolved P from areas with long term accumulation. An analysis that was done on the lag times of nutrients in the GRW found that the central basin had lower lag times as a result of dense tile drainage (Van Meter and Basu, 2017), which could be compounded by the high P erosion risk in this area. Additionally, the study found that the southern tip of the GRW had higher lag times because transport is dominantly influenced by groundwater (Van Meter and Basu, 2017). These results correlate with the idea that long lag times exist inland along the coast of Lake Erie, and perhaps this is where legacy P stores are located.

The TRW was mostly covered by accumulation risk, with some erosive risk on the northeastern edge (Figure 3-9). The Fanshawe Reservoir along the Thames River has experienced negative impacts as a result of high P loading (Nürnberg and LaZerte, 2005); the dominance of accumulation risk in the TRW suggests that the P affecting downstream waterbodies is P that had accumulated in the surrounding sediment which continuously moves towards these surface waters over time. If the accumulation risk does represent the location of legacy P, it's proximity to the shores of the LGL may be concerning.

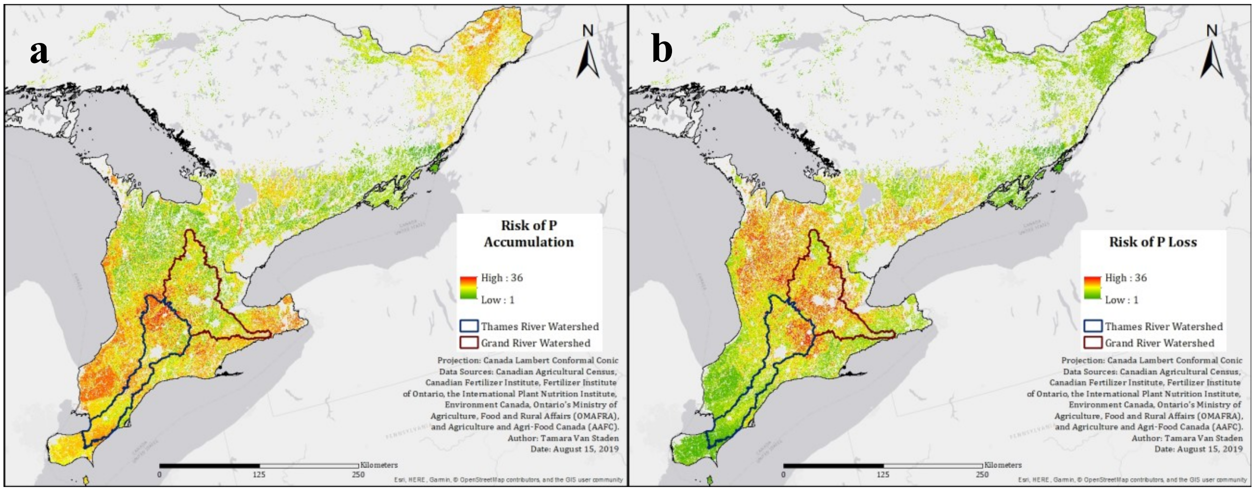


Figure 3-9. Risk of a) P loss and b) P accumulation in the major watersheds of southern Ontario.

3.3.5 P Erosion Estimation

The amount of P to have eroded from its original placement on agricultural soils was estimated using STP and RUSLE data, and ranged up to a cumulative ~1000 kg P/ha since 1981 (Figure 3-10a). Most soils had less than 10 kg/ha of potential P erosion, making it visually difficult to identify areas with high soil P erosion (Figure 3-10a). The log scale map provided a clearer visual of where the soils with higher soil P erosion potential were located (Figure 3-10b). There was a more even distribution of potential eroded soil P mass in the southwestern regions of Ontario when compared to the distribution of soil erosion vulnerability (Figure 3-10b), driven by the location of high STP concentrations and high soil erosion potential. The STP was higher along the coast of Lake Erie in southern Ontario, whereas the soil erosion potential was higher towards western Ontario. The most probable cause of this reflected distribution is that STP was higher in areas with lower erosion.

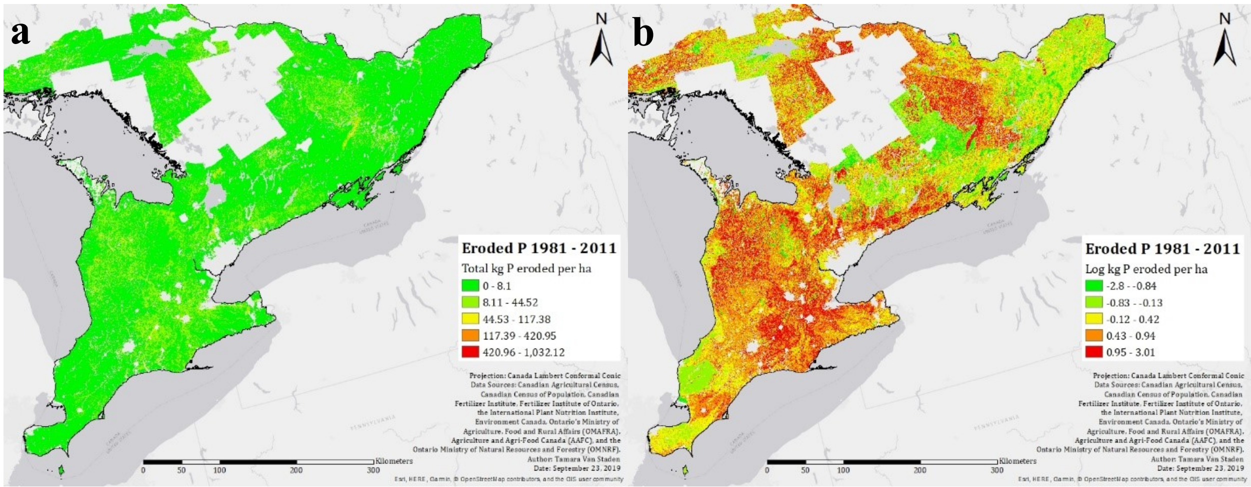


Figure 3-10. Map of estimated P erosion in the agricultural areas of Ontario from 1961 to 2016 in units a) kg P/ha and b) log(kg P/ha).

It is important to note that the estimated P mass eroded reflects conditions of maximum possible soil erosion because of the exclusion of the C and P* factors, so these erosion masses may be up to two orders of magnitude smaller depending on the dominant cropping system (i.e. hay and sod cropping systems have C and P* factors that reduce the soil erosion potential by 98% (Wall et al., 2002)). When compared to cumulative soil P input magnitudes over the same time period, some locations had higher soil P erosion than was possible. For this reason, the minimum (0.02) and maximum (0.56) C and P* factors were applied to the total soil P erosion potential to gauge a range of possible eroded soil P across the province (Table 3-10). The proportion of P erosion possible was estimated to range from 0.057 to 2.852% of the cumulative P surplus (from 1981 to 2011) estimated in Chapter 2 (Table 3-10), suggesting that > 97% of the P surplus either remained in the system, or was transported using another mechanism.

Table 3-10. Proportion of estimated cumulative P eroded over the period 1981 to 2011 using minimum and maximum C factor values.

	Minimum CP* factor Eroded P	Maximum CP* factor Eroded P	Maximum Eroded P	Cumulative surplus P
Total P (kg)	522859	14640062	26142968	916711168
Proportion of P surplus (%)	0.06	1.6	2.9	-

3.3.6 MCRA vs. RUSLE

The pilot MCRA in the Norrström basin was unique in that it provided a top-down approach to assessing the risk of P losses across the basin. Each layer was tailored through reclassification to reflect the influence on the fate and transport of P. The flexibility of the MCRA approach simplified the analysis and made it possible to visualize the influence of a combination of parameters on a large scale. Future models can also be edited to accommodate specific needs, for example, if the model is targeted towards dissolved P specifically, new layers can be selected that reflect the fate and transport of dissolved P in the subsurface, such as hydraulic conductivity.

The MCRA approach could not be used in Ontario for this project because of the larger scale and time limitations. However, a similar approach was made possible when the spatial data frame of the RUSLE for Ontario (as the layers RKLS) was made available through a collaboration with the OMAFRA. The application of the spatial RUSLE layer in lieu of a traditional vulnerability map served effectively as an intrinsic representation of the fate and transport of P as it pertains to sediment movement. It should be noted that the RUSLE model does not account for wind-based erosional processes, but it contains information about precipitation, erosive properties, and slope, all of which were considered in the MCRA analysis. The alternative approach is widely accepted as a method of estimating soil erosion potential, and the spatial data provided an excellent gauge of large-scale influences on P.

3.3.7 Sources of Error

There are often significant uncertainties associated with spatial vulnerability models due to the nature of the data. Many assumptions need to be made in conjunction with a heavy reliance on datasets that may have been interpolated. For example, in the cumulative P surplus layer it was assumed that P was distributed evenly on all cropland within the county. Other datasets were from government databases that have statements outlining the limitations of using the data. The low resolution of the dataset limits its use in fine scale studies, however, the importance of vulnerability models remains within its ability to represent potential outcomes at a large scale (Eilers and Buckley, 2002).

One method to overcome data limitations involves classifying the data into categories reflective of their environmental influence. Meaningful categorization of large datasets has the potential to improve the use of the data by decision makers (Eilers and Buckley, 2002). The RUSLE and P surplus layers were classified into qualitative groups, which was useful for informing the influence of the parameter values. Some error may be introduced by creating classes, but other types of error may be removed. For example, the precision of the values was no longer necessary, as the general influence of the value's magnitude was emphasised.

In addition, it is important to note that the RUSLE model used in the risk analysis only considered the RKLS data and did not include the C and P* parameters. Work is being conducted at OMAFRA to develop the soil management and support practices layers for the entire province by using the AAFC Crop Inventory product but could not be completed within the time frame of this study. The final vulnerability maps therefore represent the maximum potential erosion, in other words, the true intrinsic vulnerability to erosion losses or accumulation if the land surface was bare. Additionally, the inclusion of the C and P* parameters would shift the erosive potential into the lower tier reclassification categories, muting the spatial variation in the map. It was determined that there was more value in preserving the spatial differences in soil erosion potential for the vulnerability map.

3.4 Conclusions

Vulnerability analyses are strong tools for gauging large-scale spatial differences in environmental properties. They are adaptable for different contaminants and can be used for risk comparison between different regions. In this study, an alternative tool was used to estimate vulnerability to P losses in Ontario soils. The RUSLE spatial layer worked as a viable alternative to traditional vulnerability methods in the interest of time.

The final risk map signifies that there are significantly higher risks of P losses through soil erosion in western Ontario, where a significant surplus of P exists. Less erosive losses are present along the agricultural coasts of the LGL in southern Ontario, and along the southern edge of eastern Ontario. In the instance that low erosion increases the probability for P accumulation, there is a higher risk of legacy accumulation in these regions, meaning that there could be large P stores located there, especially considering that they are downgradient from high input locations. These maps are unique in that the convey the risk of P losses from, and P accumulation within the landscape in lieu of site measurement data.

Using the knowledge of risk of either erosion or accumulation, CPs can be more effectively applied. The developed risk maps can guide decision makers in addressing the large-scale influences on P transport through watersheds and help them implement adaptive conservation practices that are most appropriate to reduce P loss risks for different areas.

Chapter 4 Conclusions

4.1 Summary of Findings

The agricultural landscape of Ontario participates in nutrient cycles and the fate and transport of P. The largest concern with mismanagement of the P cycle is the HABs that occur in aquatic ecosystems that can harm wildlife and damage water quality. For the agricultural region in Ontario, the water bodies of concern are the LGL, with Lake Erie having the shallowest depth and the closest proximity to both Canadian and American agricultural P sources. Lake Erie is now a heavily studied water body because of its history with major algal blooms and the significant binational efforts to mitigate them. Today, despite successful P reduction efforts, HABs are still observed within Lake Erie, which may be attributed in part to the historical buildup of excess P in the soils and groundwater, referred to as legacy P.

In this thesis, the following objectives were set: to estimate the magnitudes of excess P in the urban and agricultural land use areas of Ontario using a large-scale long-term mass balance; to observe the differences in the mass balance elements and therefore the drivers of excess P in the north, south, east, west, and central agricultural regions in Ontario; and to use the mass balance results in conjunction with environmental conditions to gauge the risk of legacy P development and erosive P losses in agricultural land use areas of Ontario.

In Chapter 2, a mass balance was conducted for all Ontario counties for every 5 years starting in 1961 and ending in 2016. Using agricultural and population census data and average P content information from literature, livestock inputs, human waste inputs, fertilizer inputs, detergent inputs, milkhouse wastewater inputs, and crop uptake outputs were quantified and combined to estimate the annual net P surplus (or removal) in each county. The P masses were normalized to kg per hectare across their respective land use areas (agricultural or urban), converted to quaternary watershed scale and mapped. It was determined that the denser agricultural regions of southern and western Ontario had a higher P input per land use hectare due to having more fertilized area as well as ILOs that produce a significant amount of manure. The presence of ILOs created a discrepancy in P distribution, concentrating P in the center of the dense agricultural region. Surrounding counties outside of the range of manure transport must import P from other sources such as purchased synthetic fertilizer, all while the counties with ILOs are required to safely manage more manure than they need for their fields.

In Chapter 3, the results from the mass balance were used to generate two risk maps for the dense agricultural regions of Ontario with the intention of identifying locations of P accumulation and P losses through soil erosion. The mass balance results alone could not give an indication of where P would end up, only where it had entered the soil system. Additionally, modeling the fate and transport of P on such a large scale would be a feat to attempt, therefore a high-level intrinsic vulnerability and risk methodology was selected as a more manageable approach. OMAFRA applied the RUSLE methodology to the agricultural areas of Ontario to estimate maximum erosion potential on a spatial scale and shared the results as a collaboration. The erosion potential results were assumed to reflect the environmental conditions of the landscape and therefore the intrinsic vulnerability to P losses and were classified as such. It was also assumed that the inverted classification reflected the areas with intrinsic vulnerability to accumulation. After multiplying the vulnerability with the classified cumulative agricultural P input data, it was determined that P accumulation risk was higher along the coast of Lake Erie and soil P erosion risk was higher moving northwards into western Ontario. The potential for legacy P to develop in the high-risk accumulation areas close to Lake Erie are particularly concerning.

4.2 Applications and Future work

Both the risk maps and the mass balance maps have applications in large-scale decision making. The methodologies are flexible and can be applied to any region with historical agricultural data, soil texture data, precipitation data, and topography information. There are limitations to the downscaling of the results, however, the methods may be applied to any scale if the data are available for the study site. Ontario was an ideal location to conduct a mass balance and risk analysis for P considering the dense agriculture and proximity to sensitive water bodies. The results may inform governmental bodies in the region that intend to effect policies that affect nutrient cycling, particularly considering CP implementation.

4.2.1 Mass Balance Perspectives and Applications

The applications of P mass balances include guiding decision makers with visual information on a large scale, providing a more complete environmental context. Extending the NAPI analysis conducted in this study across Canada would expand the database of P inputs and would help delineate P legacies around other at-risk surface waters such as Lake Winnipeg in Manitoba. Further research involves applying the NAPI methodology to conduct a spatiotemporal mass balance of P within Canadian watersheds, beginning in Manitoba with special consideration given to the Lake Winnipeg Basin, and extending to all provinces.

The mass balance data may be used to inform mechanistic models that work at finer scales. The methods could also be used to understand finer historical trends. The quaternary watershed scale, although finer than county scale, has a low resolution when considering the needs of individual farms. If data became available for a farm-scale NAPI analysis, the resolution could be further increased, or completed on a case-by-case basis.

Detergent inputs significantly decreased over the past 55 years because of government regulations. The impact on wastewater treatment plant P-removal efficiency as a result of the most recent regulation has yet to be observed on a large scale, and could be offset by large industries, which are not required to meet the 2010 regulation standard. Another study analyzing the timing of regulations and the output from wastewater treatment plants could reflect the efficacy of government intervention on point sources.

A validation of the NAPI trends outlined in this study is being conducted by comparing agricultural P inputs to the changes in STP over time using a k-mean cluster analysis. Specifically focusing on the agricultural P balance and seeing how the change in soil P compared would be a useful gauge of the effectiveness of the NAPI model in capturing large-scale P trends. The erosion potential data provided by OMAFRA will also be included to see if the environmental conditions explain any discrepancies between agricultural P surplus and STP. Alternative validation approaches include using recent high-resolution tributary monitoring data and observing where total P losses are higher in comparison with the erosion risk maps.

4.2.2 Vulnerability and Risk Analysis Perspectives and Applications

While the map results were at a high-level resolution, their functionality is relevant in large-scale decision making. It should be noted that further validation of the maps is required before they are used to guide CP implementation, but they could be used as an initial gauge of P risk conditions to target soil testing effort in potential high P accumulation areas. Additionally, future studies can use the maps to examine the environmental conditions of their research site. MCRA analysis can be tailored to distinguish between the different P species by including parameters that influence P bioavailability (e.g. iron content) and the fate and transport (e.g. hydraulic conductivity) of P species.

Future research applications include comparing the risk maps to the current spatial distribution of CP application. Meeting reduction targets involves maximizing the efficiency of current CPs and implementing them in effective locations; comparing current CP implementation to the risk maps may outline existing inefficiencies.

The intrinsic vulnerability map generated for the Norrström basin in Sweden will be used to conduct a similar risk analysis using P data in Ontario. The vulnerability map was created with the intention of developing a methodology to estimate intrinsic vulnerability using publicly available data which should be available in most countries, and as such, the methodology could be applied in a variety of study sites. The methodology may be applied to Ontario and contrasted with the vulnerability maps that were used in this study to see how the approaches compare. In addition, the risk maps generated for the Norrström basin will be weighed against policy implementation in the region to get a sense of whether CPs have been implemented in appropriate locations.

References

Agriculture and Agri-Food Canada. (2009). Indicator of risk of water contamination by phosphorus (IROWC-P): A handbook for using the IROWC-P algorithms. (E. van Bochove, G. Theriault, and J.-T. Denault, Eds.). Quebec. https://doi.org/10.4141/cjss96-020

Allaway, C. (2003). Phosphorous loading algorithms for the South Nation River. Ottawa. Retrieved from www.nation.on.ca

Baldwin, D. J. B., Desloges, J. R., and Band, L. E. (2000). Physical Geography of Ontario. Ecology of a Managed Terrestrial Landscape: Patterns and Processes of a Forest Landscapes in Ontario. UBC Press. https://doi.org/10.1177/030913339702100208

Ballabio, C., Panagos, P., and Monatanarella, L. (2016). Mapping topsoil physical properties at European scale using the LUCAS database. Geoderma, 261, 110–123. https://doi.org/10.1016/j.geoderma.2015.07.006

Beaulieu, M. S., Bédard, F., and Lanciault, P. (2001). Distribution and Concentration of Canadian Livestock. Ottawa, ON.

Bennett, E. M., Reed-Andersen, T., Houser, J. N., Gabriel, J. R., and Carpenter, S. R. (1999). A Phosphorus Budget for the Lake Mendota Watershed. Ecosystems, 2, 69–75. https://doi.org/10.1007/s100219900059

Bocaniov, S. (2019). Discussions with Research Scientist in the Ecohydrology Research Group at the University of Waterloo.

van Bochove, E., Thériault, G., and Denault, J.-T. (2010). Indicator of risk of water contamination by phosphorus (IROWC_P). A Handbook for presenting the IROWC_P algorithms. Quebec. https://doi.org/No. A22-514/2010E-PDF

van Bochove, E., Denault, J.-T., Leclerc, M.-L., Thériault, G., Dechmi, F., Allaire, S. E., et al. (2011). Temporal trends of risk of water contamination by phosphorus from agricultural land in the Great Lakes Watersheds of Canada. Canadian Journal of Soil Science, 91(3), 443–453. https://doi.org/10.1139/CJSS09122

Booth, L., and Quinn, F. (1995). Twenty-five years of the Canada Water Act. Canadian Water Resources Journal, 20(2), 65–90. https://doi.org/10.4296/cwrj2002065

Brown, C., Follings, J., Moran, M., Rosser, B., Bagg, J., Ball, B., et al. (2017). Agronomy Guide for Field Crops. Toronto, ON. Retrieved from www.tillageontario.com

Brückmann, E. (2000). Submission by the Canadian Environmental Law Association to the Ministries of Agriculture, Food, and Rural Affairs and Environment on the Discussion Paper on Intensive Agricultural Operations in Ontario. Toronto, ON. Retrieved from http://agrienvarchive.ca/bioenergy/download/pigs_bruckman.pdf

Bruulsema, T. W., Mullen, R. W., O'Halloran, I. P., and Warncke, D. D. (2011). Agricultural phosphorus balance trends in Ontario, Michigan and Ohio. Canadian Journal of Soil Science, 91(3), 437–442. https://doi.org/10.1139/CJSS10002

Bucknell, D., Marchand, L., and McEwan, K. (2016). Farm Input Prices - Ontario and US Great Lakes Region: 2007-2016. Guelph, Ontario.

Butterbach-Bahl, K., Baggs, E. M., Dannenmann, M., Kiese, R., and Zechmeister-Boltenstern, S. (2013). Nitrous oxide emissions from soils: how well do we understand the processes and their controls? Philosophical Transactions of the Royal Society Series B, 368, 1–13. https://doi.org/10.1098/rstb.2013.0122

Canadian Fertilizer Institute. (2015). Industry Stats. Retrieved May 7, 2018, from https://fertilizercanada.ca/about-fertilizer-canada/resources/industry-resources/industry-stats/

Carpenter, S R, Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., and Smith, V. H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecological Applications, 8(3), 559–568.

Carpenter, Stephen R., Booth, E. G., and Kucharik, C. J. (2017). Extreme precipitation and phosphorus loads from two agricultural watersheds. Limnology and Oceanography. https://doi.org/10.1002/lno.10767

Casséus, L. (2009). Canadian Agriculture at a Glance: Canola: a Canadian success story.

Cheng, F. Y., and Basu, N. B. (2017). Biogeochemical hotspots: Role of small water bodies in landscape nutrient processing. Water Resources Research, 53(6), 5038–5056. https://doi.org/10.1002/2016WR020102@10.1002/(ISSN)1944-7973.HOBIES

Clegg, L. F. L. (1956). Chemical Sterilization. International Journal of Dairy Technology, 9(1), 30–35. https://doi.org/10.1111/j.1471-0307.1956.tb02883.x

Cordell, D., Drangert, J.-O., and White, S. (2009). The story of phosphorus: Global food security and food for thought. Global Environmental Change, 19(2), 292–305. https://doi.org/10.1016/J.GLOENVCHA.2008.10.009

Davis, B. (n.d.). Fertile Soil. The Noun Project.

Dodd, R. J., and Sharpley, A. N. (2016). Conservation practice effectiveness and adoption: unintended consequences and implications for sustainable phosphorus management. Nutrient Cycling in Agroecosystems, 104(3), 373–392. https://doi.org/10.1007/s10705-015-9748-8

Ebert, K., Ekstedt, K., and Jarsjo, J. (2016). GIS analysis of effects of future Baltic sea level rise on the island of Gotland, Sweden. Natural Hazards and Earth System Sciences, 16(7), 1571–1582. https://doi.org/10.5194/nhess-16-1571-2016

Eilers, R. G., and Buckley, K. E. (2002). A methodology for evaluating soils, landscapes and geology for nutrient management planning in the Prairie landscape. Retrieved from http://sis.agr.gc.ca/cansis/publications/manuals/2002-hems/hems-2002.pdf

Environment Canada. (2011). Concentration of Phosphorus in Certain Cleaning Products Regulations. Retrieved from http://publications.gc.ca/collections/collection_2011/ec/En14-51-2011-eng.pdf

Focazio, M. J., Reilly, T. E., Rupert, M. G., and Helsel, D. R. (1984). Assessing ground-Water vulnerability to contamination: providing scientifically defensible information for decision makers. Retrieved March 31, 2017, from https://pubs.usgs.gov/circ/2002/circ1224/html/new.htm

Gehrels, J., and Mulamoottil, G. (1990). Hydrologic processes in a southern Ontario wetland. Hydrobiologia, 208, 221–234. Retrieved from https://link.springer.com/content/pdf/10.1007%2FBF00007787.pdf

Government of Canada. Concentration of Phosphorus in Certain Cleaning Products Regulations, Pub. L. No. SOR/89-501 (2010). Minister of the Environment. Retrieved from https://laws-lois.justice.gc.ca/eng/regulations/SOR-89-501/index.html

Government of Canada. (2015). Great Lakes Maps. Retrieved December 12, 2017, from https://www.canada.ca/en/environment-climate-change/services/great-lakes-protection/maps.html

Government of Canada. (2018). Canada-Ontario Lake Erie Action Plan Partnering on Achieving Phosphorus Loading Reductions to Lake Erie from Canadian Sources.

Government of Ontario. Milk Act, Pub. L. No. R.S.O. 1990, c. M.12, 1 (1990). Canada. Retrieved from https://www.ontario.ca/laws/statute/90m12

Government of Ontario. Nutrient Management Act, 2002, S.O. 2002, c. 4 (2018). https://www.ontario.ca/laws/statute/02n04.

Goyette, J., Bennett, E. M., Howarth, R. W., and Maranger, R. (2016). Changes in anthropogenic nitrogen and phosphorus inputs to the St . Lawrence sub-basin over 110 years and impacts on riverine export, (October). https://doi.org/10.1002/gbc.2043

Grant, K. N., Macrae, M. L., Rezanezhad, F., and Lam, W. V. (2019). Nutrient leaching in soil affected by fertilizer application and frozen ground. Vadose Zone Journal, 18(1). https://doi.org/10.2136/vzj2018.08.0150

Great Lakes Water Quality Agreement (2012). Washington D.C., Ottawa Ontario: National Research Council of the United States and The Royal Society of Canada. Retrieved from https://binational.net//wp-content/uploads/2014/05/1094_Canada-USA-GLWQA-_e.pdf

Han, H., and Bosch, N. S. (2012). Historical pattern of phosphorus loading to Lake Erie watersheds. Journal of Great Lakes Research, 38, 289–298. https://doi.org/10.1016/j.jglr.2012.03.004

Han, H., Bosch, N., and Allan, J. D. (2011). Spatial and temporal variation in phosphorus budgets for 24 watersheds in the Lake Erie and Lake Michigan basins. Biogeochemistry, 102(1/3), 45–58.

Han, Y., Yu, X., Wang, X., Wang, Y., Tian, J., Xu, L., and Wang, C. (2013). Net anthropogenic phosphorus inputs (NAPI) index application in Mainland China. Chemosphere, 90(2), 329–337. https://doi.org/10.1016/J.CHEMOSPHERE.2012.07.023

Hasler, A. D. (1969). Cultural Eutrophication Is Reversible. BioScience, 19(5), 425–431. Retrieved from http://www.jstor.org/stable/1294478

Hilborn, D., and Stone, R. P. (2005). Determining the phosphorus index for a field. OMAFRA Fact Sheets, (05–067), 8. Retrieved from http://www.omafra.gov.on.ca/english/engineer/facts/05-067.pdf

Hjulström, F. (1935). Studies of the morphological activity of rivers as illustrated by the River Fyris. Bulletin of the Geological Institute University of Uppsala, 25, 221–527. Retrieved from http://www.fao.org/3/t1765e/t1765e0f.htm

Hofmann, N., and Beaulieu, M. S. (2006). A Geographical Profile of Manure Production in Canada, 2001. Ottawa. https://doi.org/21-601-MIE — No. 077

Holeton, C. (2013). Sources of Nutrients and Sediments in the Grand River Watershed. In Grand River Conservation Authority (Ed.), Grand River Watershed Water Management Plan (p. 73). Cambridge ON: Water Quality Working Group.

Hong, B., Swaney, D. P., Mörth, C.-M., Smedberg, E., Eriksson Hägg, H., Humborg, C., et al. (2012). Evaluating regional variation of net anthropogenic nitrogen and phosphorus inputs (NANI/NAPI), major drivers, nutrient retention pattern and management implications in the multinational areas of Baltic Sea basin. Ecological Modelling, 227, 117–135. https://doi.org/10.1016/j.ecolmodel.2011.12.002

Van Horn, H. H. (1998). Factors Affecting Manure Quantity, Quality, and Use. Gainesville. Retrieved from https://www.txanc.org/docs/vanhorn3.pdf

Howarth, R. W., Billen, G., Swaney, D., Townsend, A., Jaworski, N., Lajtha, K., et al. (1996). Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. Biogeochemistry, 35, 75–139. https://doi.org/10.1007/BF02179825

International Joint Commission. (2013). Great Lakes Water Quality Agreement. Retrieved January 5, 2018, from http://www.ijc.org/en_/Great_Lakes_Quality

International Joint Commission. (2014). A Balanced Diet for Lake Erie: Reducing Phosphorus Loadings and Harmful Algal Blooms. Retrieved from http://ijc.org/files/publications/2014 IJC LEEP REPORT.pdf

International Joint Commission. (2018). Fertilizer Application Patterns and Trends and Their Implications for Water Quality in the Western Lake Erie Basin. Retrieved from http://ijc.org/files/tinymce/uploaded/Publications/IJC_FertReport.pdf

International Plant Nutrition Institute. (2012). Reducing loss of fertilizer phosphorus to Lake Erie with the 4Rs. Retrieved from http://nane.ipni.net

International Plant Nutrition Institute. (2013a). Crop N & P Balance in Ontario, 1950-2007. Retrieved July 19, 2018, from http://phosphorus.ipni.net/article/PPP-3048

International Plant Nutrition Institute. (2013b). Eastern Canada Soil Test P. Retrieved January 16, 2019, from http://phosphorus.ipni.net/article/nane-3054

Irene. (n.d.). Lake. The Noun Project.

Jarvie, H. P., Sharpley, A. N., Withers, P. J. a., Scott, J. T., Haggard, B. E., and Neal, C. (2013). Phosphorus Mitigation to Control River Eutrophication: Murky Waters, Inconvenient Truths, and “Postnormal” Science. Journal of Environment Quality, 42, 295–304.

Jarvie, H. P., Sharpley, A. N., Spears, B., Buda, A. R., May, L., and Kleinman, P. J. A. (2013). Water quality remediation faces unprecedented challenges from “legacy phosphorus.” Environmental Science and Technology, 47, 8997–8998. https://doi.org/10.1021/es403160a

Jones, R. L. (1946). History of Agriculture in Ontario 1613-1880. University of Toronto Press. Retrieved from https://books.google.ca/books?hl=en&lr=&id=njQ5DwAAQBAJ&oi=fnd&pg=PT11&dq=history+of+agriculture+in+southern+ontario&ots=QRvtyWrK6C&sig=adXa0gIy0oEf1a2vj4yecBS3IAk#v=onepage&q=history of agriculture in southern ontario&f=false

Joosse, P. J., and Baker, D. B. (2011). Context for re-evaluating agricultural source phosphorus loadings to the Great Lakes. Canadian Journal of Soil Science, 91(3), 317–327. https://doi.org/10.1139/CJSS10005

Jordan, T. E., and Weller, D. E. (1996). Human Contributions to Terrestrial Nitrogen Flux. Source: BioScience, 46(9), 655–664. Retrieved from http://www.jstor.org/stable/1312895

Kellogg, R. L., Lander, C. H., Moffitt, D. C., and Gollehon, N. (2000). Manure Nutrients Relative to the Capacity of Cropland and Pastureland to Assimilate Nutrients: Spatial and Temporal Trends for the United States. Retrieved from www.nhq.nrcs.usda.gov/land/index/publication.html

Kirchmann, H., and Thorvaldsson, G. (2000). Challenging targets for future agriculture. European Journal of Agronomy, 12, 145–161. https://doi.org/10.1016/S1161-0301(99)00053-2

Lagunov, D. (n.d.). Crystal. The Noun Project.

Lander, C. H., Moffitt, D. C., and Alt, K. (1998). Nutrients Available from Livestock Manure

Relative to Crop Growth Requirements. Retrieved from https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/technical/nra/?cid=nrcs143_014175

LimnoTech. (2017). Assessment of Fertilizer and Manure Application in the Western Lake Erie Basin. Retrieved from http://ijc.org/files/publications/LimnoTech_IJC_Fertilizer.pdf

Litke, D. W. (1999). Review of Phosphorus Control Measures in the United States and Their Effects on Water Quality. Denver, Colorado.

Long, C. M., Logsdon Muenich, R., Kalcic, M. M., and Scavia, D. (2018). Use of manure nutrients from concentrated animal feeding operations. Journal of Great Lakes Research, In Press. https://doi.org/10.1016/j.jglr.2018.01.006

Loomer, H. A., and Cooke, S. E. (2011). Water Quality in the Grand River Watershed: Current Conditions & Trends (2003 – 2008) October 2011.

Maccoux, M. J., Dove, A., Backus, S. M., and Dolan, D. M. (2016). Total and soluble reactive phosphorus loadings to Lake Erie A detailed accounting by year, basin, country, and tributary. Journal of Great Lakes Research, 42, 1151–1165. https://doi.org/10.1016/j.jglr.2016.08.005

MacDonald, G. K., and Bennett, E. M. (2009). Phosphorus accumulation in saint lawrence river watershed soils: A century-long perspective. Ecosystems, 12(4), 621–635. https://doi.org/10.1007/s10021-009-9246-4

Maki, A. W., Porcella, D. B., and Wendt, R. H. (1984). The impact of detergent phosphorus bans on receiving water quality. Water Research, 18(7), 893–903.

McGucken, W. (1989). The Canadian Federal Government, Cultural Eutrophication, and the Regulation of Detergent Phosphates, 1970. Environmental Review: ER, 13(3/4), 155–166. Retrieved from https://www.jstor.org/stable/pdf/3984394.pdf?refreqid=excelsior%3A16b0d68b4df4b680a214be1464e53708

McKague, K. (2019). Discussions with the Erosion Engineer at the Ontario Ministry of Agriculture, Food and Rural Affairs. 2nd July.

Van Meter, K. J., and Basu, N. B. (2015). Catchment legacies and time lags: A parsimonious watershed model to predict the effects of legacy storage on nitrogen export. PLoS ONE, 10(5), 1–22. https://doi.org/10.1371/journal.pone.0125971

Van Meter, K. J., Basu, N. B., Veenstra, J. J., and Burras, C. L. (2016). The nitrogen legacy: emerging evidence of nitrogen accumulation in anthropogenic landscapes. Environmental Research Letters, 11(3), 1–12. https://doi.org/10.1088/1748-9326/11/3/035014

Van Meter, K. J., Basu, N. B., and Van Cappellen, P. (2017). Two centuries of nitrogen dynamics: Legacy sources and sinks in the Mississippi and Susquehanna River Basins. Global Biogeochemical Cycles, 31(1), 2–23. https://doi.org/10.1002/2016GB005498

Van Meter, Kim J, and Basu, N. B. (2017). Time lags in watershed-scale nutrient transport: an exploration of dominant controls. Environmental Research Letters, 12. https://doi.org/10.1088/1748-9326/aa7bf4

MFRA. (n.d.). asbestos. The Noun Project.

Michalak, A. M., Anderson, E. J., Beletsky, D., Boland, S., Bosch, N. S., Bridgeman, T. B., et al. (2013). Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. Proceedings of the National Academy of

Sciences of the United States of America, 110(16), 6448–6452. https://doi.org/10.1073/pnas.1216006110

Munroe, J., Brown, C., Kessel, C., Verhallen, A., Lauzon, J., O'Halloran, I., et al. (2018). Soil Fertility Handbook. https://doi.org/611

Neff, B. P., Piggott, A. R., and Sheets, R. A. (2005). Estimation of Shallow Ground-Water Recharge in the Great Lakes Basin. Reston Virginia. Retrieved from https://pubs.usgs.gov/sir/2005/5284/pdf/SIR_2005_5284-Web.pdf

Nürnberg, G. K., and LaZerte, B. D. (2005). Reservoir Water Quality Treatment Study Includes Water Quality Assessment and Modeling for the reservoirs Fanshawe Lake, Wildwood, and Pittock. Baysville, Ontario. Retrieved from http://thamesriver.on.ca/wp-content/uploads/SurfaceWater/ReservoirWaterQualityTreatmentStudy1-GertrudNurnberg.pdf

OMAFRA. (2001). Ontario Crop Report - 2001 Seasonal Summary. Retrieved November 21, 2018, from http://www.omafra.gov.on.ca/english/crops/field/reports/2001summary.htm

OMAFRA. (2016). Agriculture in the Great Lakes Basin – Stewardship and Innovation. Retrieved December 12, 2017, from http://www.omafra.gov.on.ca/english/environment/facts/gl_basin.htm

OMAFRA. (2018). Field Crops. Retrieved from http://www.omafra.gov.on.ca/english/stats/crops/index.html

Ontario Federation of Agriculture. (2011). Environmental Farm Plans: Measuring Performance, Improving Effectiveness, and Increasing Participation. Retrieved from www.pra.ca

Ontario Ministry of Natural Resources and Forestry - Provincial Mapping Unit. (2010). Quaternary Watershed. Retrieved December 29, 2017, from https://www.javacoeapp.lrc.gov.on.ca/geonetwork/srv/en/main.home?uuid=7a99025f-b894-4b8c-97c4-60f830fa1acc

Orihel, D. M., Baulch, H. M., Casson, N. J., North, R. L., Parsons, C. T., Seckar, D. C. M., and Venkiteswaran, J. J. (2017). Internal phosphorus loading in Canadian fresh waters: a critical review and data analysis. Canadian Journal of Fisheries and Aquatic Sciences, 2029(September), 1–25. https://doi.org/10.1139/cjfas-2016-0500

Oryschak, M., Chrapko, D., Huggins-Rawlins, N., Jones, S., Korth, C., Kryzanowski, L., et al. (2008). Nutrient Management Planning Guide Forward Nutrient Management Planning Guide for Alberta. Edmonton. Retrieved from https://www1.agric.gov.ab.ca/$department/deptdocs.nsf/all/epw11920/$FILE/nutrient-management-planning-guide.pdf

Paerl, H. W. (1988). Nuisance phytoplankton blooms in coastal, estuarine, and inland waters. Limnology and Oceanography, 33(4, part 2), 823–847. https://doi.org/10.4319/lo.1988.33.4part2.0823

Pardy, A. (2019). Then and Now: Understanding Change on an Agricultural Landscape. Ryerson University.

PLUARG. (1974). Pollution from Land Use Activities Reference Group. Retrieved December 12, 2019, from https://atrium.lib.uoguelph.ca/xmlui/handle/10214/13066

Plummer, R., Spiers, A., Summer, R., and FitzGibbon, J. (2007). The Contributions of Stewardship to Managing Agro-Ecosystem Environments. Journal of Sustainable Agriculture, 31(3), 55–84. https://doi.org/10.1300/J064v31n03_06

Powers, S. M., Bruulsema, T. W., Burt, T. P., Chan, N. I., Elser, J. J., Haygarth, P. M., et al. (2016). Long-term accumulation and transport of anthropogenic phosphorus in three river basins. Nature Geoscience, 9(5), 353–356. https://doi.org/10.1038/ngeo2693

Reddy, K. R., Kadlec, R. H., Flaig, E., and Gale, P. M. (1999). Phosphorus Retention in Streams and Wetlands: A Review. Critical Reviews in Environmental Science and Technology, 29(1), 83–146. https://doi.org/10.1080/10643389991259182

Reid, K., Schneider, K., and McConkey, B. (2018). Components of Phosphorus Loss From Agricultural Landscapes, and How to Incorporate Them Into Risk Assessment Tools. Frontiers in Earth Science, 6, 135. https://doi.org/10.3389/feart.2018.00135

Robertson, D. M., and Saad, D. A. (2011). Nutrient Inputs to the Laurentian Great Lakes by Source and Watershed Estimated Using SPARROW Watershed Models. Journal of the American Water Resources Association, 47(5), 1011–1033. https://doi.org/10.1111/jawr.12060

Russell, M. J., Weller, D. E., Jordan, T. E., Sigwart, K. J., Sullivan, K. J., Biogeochemistry, S., et al. (2008). Net Anthropogenic Phosphorus Inputs : Spatial and Temporal Variability in the Chesapeake Bay Region. Biogeochemistry, 88(3), 285–304. https://doi.org/10.1007/sl0533-008-9212-9

Samson, M.-I. (2019). The Urban Metabolism of the Greater Toronto Area: A Study of Nitrogen and Phosphorus Fluxes across the Urban, Suburban, and Rural Continuum. University of Waterloo. Retrieved from https://uwspace.uwaterloo.ca/bitstream/handle/10012/14658/Samson_Melani-Ivy.pdf?sequence=4&isAllowed=y

Saurette, D. (2015). Soil Survey Complex. Guelph Ontario: Ontario Ministry of Agriculture, Food and Rural Affairs. Retrieved from https://www.javacoeapp.lrc.gov.on.ca/geonetwork/srv/en/main.home?uuid=86302406-ddff-4505-b3af-39c293a6702a

Schindler, D. W. (1977). Evolution of Phosphorus Limitation in Lakes. Science, 195(4275), 260–262. Retrieved from http://links.jstor.org/sici?sici=0036-8075%2819770121%293%3A195%3A4275%3C260%3AEOPLIL%3E2.0.CO%3B2-N

Sharpley, A., Daniel, T., Gibson, G., Bundy, L., Cabrera, M. L., Sims, T., et al. (2006). Best Management Practices to Minimize Agricultural Phosphorus Impacts on Water Quality.

Sharpley, A., Jarvie, H. P., Buda, A., May, L., Spears, B., and Kleinman, P. (2013). Phosphorus Legacy: Overcoming the Effects of Past Management Practices to Mitigate Future Water Quality Impairment. Journal of Environmental Quality, 42(5), 1308–1326. https://doi.org/10.2134/jeq2013.03.0098

Shifflett, S., Boyd, D., and Etienne, J. (2012). Limitations to the change field method and future considerations for climate change investigations in the Grand River watershed. In Canadian Dam Association Meeting. Saskatoon, Saskatchewan.

Shigaki, F., Sharpley, A., and Prochnow, L. I. (2006). Animal-Based Agriculture, Phosphorus Management and Water Quality in Brazil: Options for the Future. Scientia Agricola, 63(2), 194–209. https://doi.org/10.1590/S0103-90162006000200013

Sims, J. T., and Kleinman, P. J. A. (2005). Managing agricultural phosphorus for environmental protection. In J. T. Sims and A. N. Sharpley (Eds.), Phosphorus: Agriculture and the Environment (pp. 1021–1068). Madison, W.I: Soil Science Society of America Special Publication. Retrieved from

https://www.ars.usda.gov/research/publications/publication/?seqNo115=174555
Smith, P. (2015). Long-Term Temporal Trends in Agri-Environment and Agricultural Land Use in Ontario, Canada: Transformation, Transition and Significance. Journal of Geography and Geology, 7(2), 32–55. https://doi.org/10.5539/jgg.v7n2p32
Smith, V. H., and Schindler, D. W. (2009). Eutrophication science: where do we go from here? Trends in Ecology and Evolution, 24(4), 201–207. https://doi.org/10.1016/j.tree.2008.11.009
Van Staden, T., Cey, E., Ryan, C., and Checkley, S. (2019). Assessing and Mapping Groundwater Vulnerability to Bacteria in Alberta. Journal of Undergraduate Research in Alberta, 7, 30–48.
Statistics Canada. (2003). Food expenditure in Canada. Ottawa. https://doi.org/62-554-XIE
Statistics Canada. (2009). Food Statistics. Statistics Canada, Agriculture Division. Ottawa ON. https://doi.org/Catalogue no. 21-020-X
Statistics Canada. (2018). Estimated areas, yield, production, average farm price and total farm value of principal field crops, in metric and imperial units. Retrieved from https://www150.statcan.gc.ca/t1/tbl1/en/tv.action?pid=3210035901
Stone, M., and Saunderson, H. (1992). Particle size characteristics of suspended sediment in southern Ontario rivers tributary to the Great Lakes. Hydrological Processes, 6, 189–198. Retrieved from https://onlinelibrary.wiley.com/doi/pdf/10.1002/hyp.3360060207
Sugg, Z. (2007). Assessing U.S. Farm Drainage: Can GIS Lead to Better Estimates of Subsurface Drainage Extent? Agriculture, (August).
Tan, C. S., and Reynolds, W. D. (2003). Impacts of Recent Climate Trends on Agriculture in Southwestern Ontario. Canadian Water Resources Journal, 28(1), 87–97. https://doi.org/10.4296/cwrj2801087
Vickers, S. (2017). Development and Application of a Soil Phosphorus Sensitivity Map for a Lake Simcoe Watershed. University of Guelph.
Wall, G. J., Coote, D. R., Pringle, E. A., and Shelton, I. J. (2002). RUSLEFAC - Revised Universal Soil Loss Equation for Application in Canada A Handbook for Estimating Soil Loss from Water Erosion in Canada. Ottawa, Ontario. https://doi.org/AAFC/AAC2244E
Water Management Plan Project Team. (2014). Grand River Watershed Management Plan. Cambridge ON.
Widory, D., Kloppmann, W., Chery, L., Bonnin, J., Rochdi, H., and Guinamant, J. L. (2004). Nitrate in groundwater: An isotopic multi-tracer approach. Journal of Contaminant Hydrology, 72, 165–188. https://doi.org/10.1016/j.jconhyd.2003.10.010
Winter, J. G., and Duthie, H. C. (2000). Export coefficient modeling to assess phosphorus loading in an urban watershed. Journal of the American Water Resources Association, 36(5), 1053–1061. https://doi.org/10.1111/j.1752-1688.2000.tb05709.x
Woyzbun, E. (2012). Spatial Analysis of the Adoption of Nutrient Management Related Best Management Practices in Ontario (April 2005 - March 2010). Retrieved from https://www.ontariosoilcrop.org/wp-content/uploads/2015/08/final_report_spatial_analysis_nm_bmp_ontario.pdf
Zhang, X. (2016). Spatio-temporal patterns in net anthropogenic nitrogen and phosphorus inputs across the Grand River Watershed. MSc. Thesis University of Waterloo.

Appendix A

Table A-1. Annual crop yields for the major crops in Ontario in tonnes per hectare (tonnes/ha). Data from OMAFRA and Statistics Canada.

Year	1961	1966	1971	1976	1981	1986	1991	1996	2001	2006	2011	2016
Winter Wheat	2.05	2.37	2.70	3.02	3.50	3.70	3.60	2.70	4.80	5.60	5.04	6.10
Spring Wheat	1.91	2.07	2.24	2.40	2.70	2.80	2.30	2.80	3.50	3.40	3.50	3.60
Buckwheat	0.51	0.65	0.78	0.92	1.10	0.90	0.90	1.50	1.90	1.75	1.89	2.03
Rye	1.89	1.95	2.02	2.08	2.30	2.20	2.10	2.00	2.20	2.30	2.60	3.00
Oats	1.58	1.69	1.81	1.93	2.00	2.30	2.10	2.00	2.60	2.50	2.70	3.20
Barley	2.80	2.86	2.92	2.98	3.10	3.20	2.90	3.00	3.60	3.40	3.30	3.40
Mixed Grain	2.45	2.51	2.57	2.62	2.80	2.80	2.60	2.60	3.10	3.00	2.90	3.00
Grain Corn	2.87	3.53	4.18	4.84	6.00	6.30	6.90	7.00	6.50	9.40	9.50	9.90
Canola	1.27	1.36	1.45	1.55	1.64	2.00	1.70	2.20	2.20	2.30	2.10	2.30
Soybeans	1.75	1.86	1.98	2.09	2.20	2.50	2.40	2.50	1.40	3.10	3.20	3.10
Flaxseed	0.44	0.65	0.85	1.06	1.60	1.40	2.20	1.90	2.11	2.32	2.53	2.74
White Beans	0.83	0.97	1.12	1.26	1.45	0.95	2.26	1.49	1.47	2.29	2.20	2.20
Coloured Beans	0.59	0.75	0.91	1.06	1.22	1.38	1.54	1.55	1.10	2.20	2.20	1.90
Fodder Corn	13.74	16.35	18.96	21.57	29.90	28.60	27.40	27.30	23.90	38.40	39.01	45.79
Hay	6.85	6.86	6.87	6.88	6.80	7.60	6.90	5.70	4.60	6.00	5.60	5.39
Potato	19.52	19.88	20.24	20.59	20.95	22.88	20.18	22.42	20.74	23.43	19.62	21.30
Tobacco	1.74	1.79	1.84	1.89	1.94	1.99	2.03	2.08	2.25	1.97	2.23	2.28

Table A-2. Laundry and dish detergent use per capita for the years of interest for the model, modified from Han and Bosch (2012) and Zhang (2016).

	Laundry Detergent Consumption P_{DL} (kg P/capita/year)	**Dish Detergent Consumption** P_{DD} (kg P/capita/year)
1901 - 1931	0	0
1941	0.099	0.005
1951	0.241	0.014
1961	0.531	0.062
1966	0.849	0.103
1971	0.542	0.139
1976	0.145	0.209
1981	0.147	0.242
1986	0.147	0.235
1991	0.147	0.227
1996	0.147	0.226
2001	0.147	0.227
2006	0.147	0.227
2011	0.033	0.018
2016	0.033	0.018

Table A-3. Data sources of parameter components from the NAPI model.

NAPI Parameter	Parameter Component	Data Source	Spatial Scale	Time Scale
Detergent	Per capita detergent use	(Han and Bosch, 2012)	Provincial	1961 – 2016
	Detergent regulations	(Government of Canada, 2010)	Provincial	2010 – 2016
	Population size	Canadian Census of Population	County	1961 – 2016
Fertilizer	Fertilizer Sales	(International Plant Nutrition Institute, 2013a)	Provincial	1954 – 2007
		(Canadian Fertilizer Institute, 2015)	Provincial	2008 – 2016
	Fertilized area	Canadian Agricultural Census	County	1961 – 2016
Human consumption	Available P	(Statistics Canada, 2009)	Provincial	1981 – 2009
	Population size	Canadian Census of Population	County	1961 – 2016
Crop production	Crop area	Canadian Agricultural Census	County	1961 – 2016
	Crop yields	(Statistics Canada, 2018)	Provincial	1961 – 1981
		(OMAFRA, 2018)	Provincial	1986 – 2016
	Crop P content	Canola: (Hong et al., 2012) All others: (Zhang, 2016)	Provincial	Constant
Livestock consumption and production	Livestock numbers	Canadian Agricultural Census	County	1961 – 2016
	Intake values	Horses: Estimated from (Y. Han et al., 2013) All others: Estimated from (Hofmann and Beaulieu, 2006)	Provincial	Constant
	Excretion values	Horses: (Y. Han et al., 2013) All others: (Hofmann and Beaulieu, 2006)	Provincial	Constant
Milk house waste	Number of dairy cows	Canadian Agricultural Census	County	1961 – 2016
	P input per head	(Allaway, 2003)	Provincial	1991, linearly extrapolated to 1941

Appendix B

Table B-1. Surficial geology classes from the Geological Survey of Sweden, translations, and VIs.

Original	Translation	Vulnerability Index
Älvsediment, grovsilt-finsand	River sediment, coarse silt, fine sand	3
Älvsediment grus	River sediment, Gravel	0
Älvsediment ler-silt	River sediment, clay/silt	1.5
Älvsediment sand	River sediment, sand	2
Berg	Exposed bedrock	0
Blockmark	Exposed bedrock	0
Fanerozoisk diabas	Exposed bedrock	0
Flygsand	Shifting sand	2.5
Flytjord eller skredjord	Silty landslide material	2
Fyllning	Coarse fill material	0
Glacial grovlera	Coarse clay	1.5
Glacial grovsilt-finsand	Coarse silt-fine sand	3
Glacial lera	Clay	1
Glacial silt	Silt	2
Grusig morän	Gravelly moraine	0
Gyttja	Mud	1.5
Gyttjelera (eller lergyttja)	Mudstone	1.5
Isälvsediment	Glacial river sediment	2
Isälvsediment grus	Glacial river gravel	0
Isälvsediment sand	Glacial river sand	2
Isälvsediment sten-block	Glacial river boulders	0
Kämtorv	Marsh peat	0
Lera	Clay	1
Lera-silt	Clay silt	1.5
Lera-silt, tifvis under vatten	Underwater clay silt	1.5
Lerig morän	Muddy moraine, more clay and silt	1.5
Morän	Moraine	1
Morän omväxlande med sort	Poorly sorted moraine	1
Morän sand	Moraine sand	2
Morän grovlera	Coarse moraine, coarse sand and gravel	1
Moränlera	Boulder clay, bimodal	1
Moränlera eller lerig morän	Moraine or muddy moraine	1
Mossetorv	Peat moss	0
Postglacial finlera	Fine clay	1
Postglacial findsand	Fine sand	3
Postglacial grovlera	Coarse clay	1.5
Postglacial grovsilt-findsand	Corse silt to fine sand	3
Postglacial lera	Clay	1
Postglacial sand	Sand	2
Postglacial silt	Silt	2
Rösberg	Red rock	0
Sandig morän	Sandy moraine	2
Sandig-siltig morän	Sandy-silty moraine	3
Sedimentärt berg	Sedimentary rock	0
Silt	Silt	2
Skaljord	Shells, gravel and sand	1
Svallsediment, grus	Gravel	0
Svämsediment grovsilt—finsand	Swimming sediment coarse silt – fine sand	3
Svämsediment ler-silt	Clay silt	1.5
Svämsediment sand	Sand, assuming medium grain	2
Talus (rasmassor)	Talus rubble	0
Torv	Peat	0
Torv, tidvis under vatten	Peat under water	0
Urberg	bedrock	0
Vittringsjord	Weathering soil	3
Vittringsjord, sand-grus	Weathering soil, sand gravel	2