Ecosystem Services: Linking ecohydrology with economic valuation

by

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Examining Committee Membership

The following served on the Examining Committee for this thesis. The decision of the Examining Committee is by majority vote.

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AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this 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

The chapters included in this dissertation are co-authored and I am the first author on all the chapters. Therefore, the responsibility of designs and execution of studies lies with me. The contributions of co-authors of each chapter are summarized below.

Chapter 2

Philippe Van Cappellen (PVC) and I designed the study. I developed the methodology, analyzed data and performed the calculations with inputs from PVC. Roy Brouwer (RB) provided useful feedback on the overall chapter and I improved the results in the light of his comments. I wrote the chapter; PVC, Hans Dürr (HD) and Erin Jones (EJ) helped me with the write up.

Chapter 3

I designed this study capitalizing on the idea of PVC. I developed and implemented a framework to achieve the required results. PVC and I wrote the chapter.

Chapter 4

After discussions with PVC and RB, I designed this study, gathered and analyzed the data and performed calculations. I wrote this chapter.

Chapter 5

To research PVC's intriguing idea, I designed this study. I collected data, used a model and created maps. Finally, I wrote the chapter.

Abstract

Economic valuation of ecosystem services has become a dominant model for environmental management at local, regional and global scales. However, policy-makers at all scales take these value estimates with a pinch of salt. Their concerns are the uncertainties accompanying value estimates, which arise from a wide variety of methods and datasets involved, underlying assumptions to capture complex ecosystem processes, and use of less accurate valuation methods in the data-scarce regions. These challenges call for bracing up the valuation methodology to yield sufficiently rational, scientifically valid and politically-acceptable estimates. This thesis, on the one hand, addresses methodological inconsistencies in the valuation approach and, on the other, develops and demonstrates the techniques to make valuation results more appropriate for incorporation into decision-making processes.

The first chapter of my thesis redefines ecosystem services, reviews valuation methods, and poses research questions. In Chapter 2, I present a comprehensive methodology for valuation of ecosystem services at watershed scale, and apply it to assess the value of four ecosystem services in response to long term land use changes in the Grand River watershed, Ontario, Canada. Unlike existing valuations of watersheds, my methodology takes into account the traditionally unvalued ecosystem services from agricultural land uses. The results show a decline in the total value of ecosystem services due to agricultural expansion, but that reforestation helps regain some of the lost value. To emphasize the use of different economic methods for valuation of consumptive and non-consumptive services, I demonstrate their different responses to the land use change in the watershed. My results suggest that locally-relevant unit values significantly reduce the variation in the total value of the watershed.

In Chapter 3, I establish a framework to distinguish the value of ecosystem services provided by different wetland types. Using this framework, I develop wetland value functions for water filtration service and apply these to four major wetland types present in southern Ontario. The results of this study show that fens are the least valued type for water filtration; a bog, a marsh and a swamp are 1.72, 2.66 and 1.56 times more valuable, respectively, than an equal size fen. Further, the cost-effectiveness analysis for phosphorus removal shows that human-made infrastructures are very costly options to replace these wetlands.

Chapter 4 determines the veracity of value estimates that are based on the value transfer method and different datasets. I use two global, one regional, and one local dataset on unit values ($/ha/year); the local

dataset serves as a baseline. The findings show that the regional dataset gives a better estimate than the global datasets. Therefore, this study recommends developing and using regional datasets to better influence policy-making. In this chapter, I also assess the impact of land use resolution on the total value of a watershed. The results indicate that a higher resolution of land use data results in a higher value and vice-versa.

In Chapter 5, I use a phenomenological model — Co$ting Nature — to capture the realized ecosystem services in southern Ontario, Canada. This model maps realized ecosystem services as scalar indices between 0 and 1. I rescale these indices locally and conform them for use in economic valuation. My results show that the value of realized ecosystem services is 50% of the value of potential ecosystem services in the selected region. Additionally, the resulting map can guide future investments in natural infrastructure to locate hotspots that matter for human well-being. Finally, Chapter 6 concludes research presented in this thesis, and sets directions for future research.

Acknowledgements

The PhD is a trip of a lifetime and this trip for me would have been more difficult without the help, support and encouragement of quite a number of people.

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And, lastly but most importantly, I would like to thank my wife, Afeera, for all her support throughout our time together during my PhD. Across these years, I went through thick and thin and her constant support has always been there at every step of the way.

Dedication

To my parents, who managed me a lantern that lit my way, shipped me to a metropolis that transformed me into a global citizen.

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Chapter 1
Introduction

Natural ecosystems and their processes are preconditions for human existence, survival and well-being. Ecosystems are the capital stocks that produce ecosystem services which are critical for human welfare, both directly and indirectly (Costanza et al., 1997). According to the Millennium Ecosystem Assessment Report (2005), ecosystems are the dynamic complexes of living (e.g., plants, animals, microorganisms) and non-living (e.g., soil, water, air) environments, of which humans are an integral part (François et al., 2005). In this human-dominated era – the Anthropocene – ecosystems are rapidly changing and degrading due to high-impact alterations such as deforestation and drainage of wetlands for agriculture (Morse et al., 2014; Rayome, 2015). Many anthropogenic changes made to the ecosystems are difficult or impossible to revert (Collier and Devitt, 2016). It is, therefore, vital to predict the response of ecosystems to the drivers of change for their better management. At the same time, our economic system fails to fully factor in the contribution of these ecosystems towards human well-being; thus, creating an efficient marketplace to demonstrate the true values of natural ecosystems will be an important step towards improving their protection. This thesis develops and refines methodologies for monetary assessment of ecosystem services.

1.1 Ecosystem services

The term "ecosystem services" appeared in the early 1980s to help explain the biophysical nature of ecosystem processes and their relation to human well-being. With modernity has come the promise that technology can provide ecosystem services more efficiently and reliably. The reality is that technology works efficiently in complementarities with environmental services. For example, while water treatment plants can bring water quality to the mandated levels, they may do so more efficiently in the presence of environmental filtration (Brauman et al., 2007). Similarly, some energy efficient buildings in Germany have been ranked worse than a dirty car-clogged street due to poor air circulation. Further, reduced air circulation in buildings is a cause of allergies to 42% of 7-year old children in Germany (McDonough and Braungart, 2003).

Traditionally, ecosystem services have been defined in various publications as the benefits that people receive from nature (Brauman et al., 2007; François et al., 2005). Some definitions include the direct and indirect benefits derived by people (Costanza et al., 1997), that is, the components of nature directly consumed and enjoyed by people (Boyd and Banzhaf, 2007), but also those aspects of ecosystems utilized for human well-being (Fisher et al., 2008), and the direct and indirect contributions of ecosystems to human well-being (Braat and de Groot, 2012; TEEB, 2010). These definitions share the idea that an ecosystem service must inevitably benefit people. In fact, even in the absence of people, ecosystems continue to produce ecosystem services. Likewise, in the ecosystem science literature, production of goods and services by ecosystems have been considered as ecosystem services regardless of their use by the people. For segregating used and unused portions, I have modified these definitions and redefined ecosystem services as the benefits from ecosystems that are at the disposal of people to use and benefit from (i.e., not the benefits that people receive). The used plus the unused benefits are called the ***potential ecosystem services*** whereas the used benefits only are called the ***realized ecosystem services*** (Mulligan and Clifford, 2015).

Other contemporary researchers have also questioned the traditional definitions of ecosystem services. For instance, La Notte et al. (2017) agree that ecosystem services (such as biomass production) are not the benefits; in fact, the benefits are the outputs of the ecosystem services. Further, they argue that some processes such as nutrient cycling are mistaken as ecosystem services (La Notte et al., 2017). In other words, there is no single or fixed definition of ecosystem services, indicating that the concept, and its practical and theoretical applications, are still an evolving topic.

Ecosystem services offer a lens for viewing human-environment relationships. An ecosystem service has multiple facets, and its assessment requires expertise in many disciplines such as ecology, biology, economics, anthropology, etc. Accordingly, the ecosystem service approach allows the bringing together of knowledge from different sectors and disciplines into a single conceptual framework. But it also means that multidisciplinary teams are needed to make the ecosystem service approach work (Smith et al., 2013).

Ecosystems possess a potential of supplying ecosystem services based on their structure and functioning. The flow of ecosystem services depend on the land cover (such as, forest and wetlands) and the ecosystem functions they express, as well as additional inputs, including financial, human, and social inputs. The demands of society distinguishes the potential ecosystem services from the used ecosystem services (Figure 1.1) (Burkhard et al., 2014).

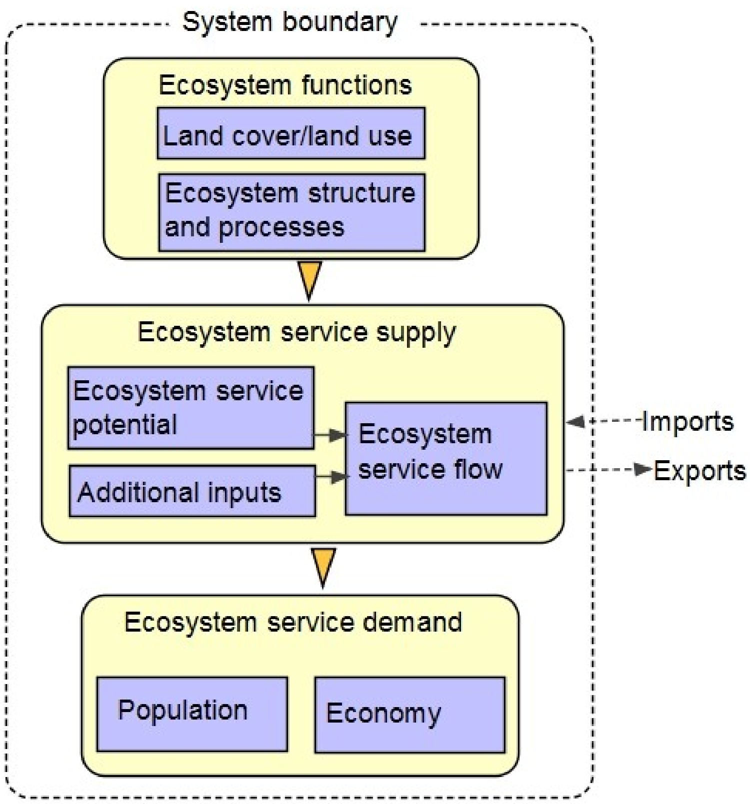

Figure 1.1: Relationship between ecosystem functions, services and benefits. Modified from Burkhard et al. (2014).

1.1.1 Classification of ecosystem services

In the broadest terms, ecosystem services are classified into four major types: 1) provisioning – products gained from ecosystems (e.g., food, fiber, genetic resources, freshwater, etc.); 2) regulating – benefits obtained from regulation processes (e.g., air quality regulation, water regulation, pollination, etc.); 3) cultural – nonmaterial benefits from ecosystems (e.g., aesthetic value, educational value, religious value, etc.) and 4) supporting – services that support production of other ecosystem services (e.g., soil formation, primary production, nutrient cycling, etc.) (Alcamo et al., 2005). Many authors have recognized that ecosystem processes (means) and services (ends) are mixed in the ecosystem literature (e.g., Costanza et al., 1997; de Groot et al., 2002). With few exceptions, the regulating and supporting services, in fact, are processes (Figure 1.2). However, Wallace (2007) recommended that a coherent classification system can be developed by addressing the linguistic uncertainty in the key terms such as ecosystem functions, processes, and services (Wallace, 2007).

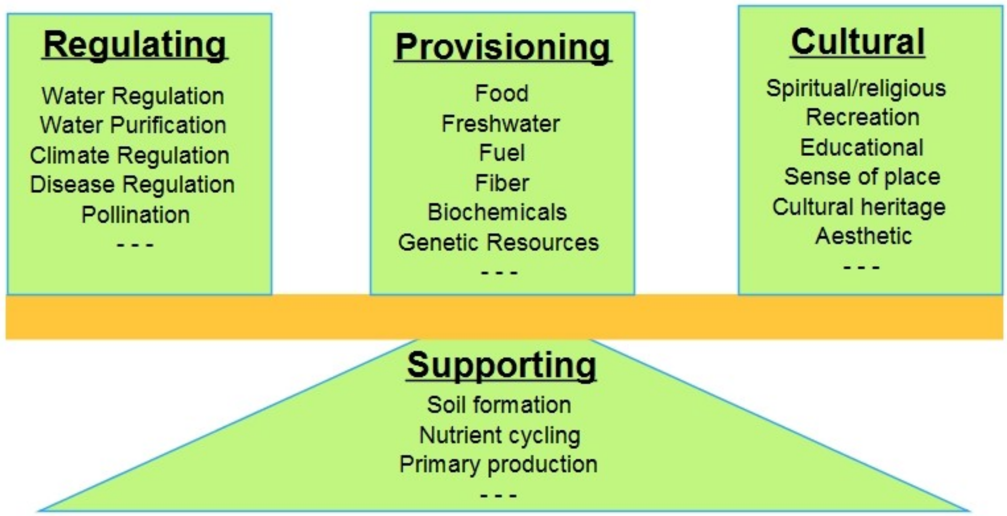

Figure 1.2: Functional classification of ecosystem services (redrawn after François et al., 2005).

1.1.2 Human well-being and ecosystem services

Ecosystem services are vital for human well-being and sustainable development. By human well-being, I am speaking broadly of the fulfillment of human needs, including psychological, spiritual and survival needs. Similarly, the Millennium Ecosystem Assessment Report (2005) links human well-being to good health, sound social relations, freedom of choice, security and access to basic goods such as food, water, etc. (Wu, 2013). Ecosystem services play a critical role in advancing human well-being by helping people meet their psychological, spiritual and survival needs. The need for reliable information on all ecosystem services categories towards sustainable development is emphasized in many policy documents, for example, the sustainable development goals (SDGs) set by United Nations (UN) and the Convention on Biological Diversity (CBD) Aichi Targets. Out of total 17 SDGs and 20 Aichi Targets, 12 goals and 17 targets are directly concerned with ecosystem services. For instance, hazard regulation is mentioned in SDG 13 and carbon sequestration is referred in Aichi Target 15. Therefore, ecosystem services contribute directly and indirectly to human well-being as shown in Figure 1.3 (Geijzendorffer et al., 2017).

The provision of ecosystem services also contributes to poverty alleviation because the livelihood of the poor is directly dependent on ecosystem services (Suich et al., 2015). Cash income and employment are the direct contributions of ecosystem services to poor communities (Daw et al., 2011). The reviews of ecosystem services studies in the developing world suggest a strong correlation between negative ecosystem services impact and negative poverty impact, and positive ecosystem services impact and positive poverty impact (Suich et al., 2015). That is, when the magnitude and quality of ecosystem services go down, poverty often increases, while when they go up, poverty be alleviated.

Nature makes direct contributions to human production systems (e.g., businesses) in terms of ecosystem services at various stages of the production process (EPA, 2015). These contributions also need human/additional inputs for their inclusion into the production system (Figure 1.1). There exists, however, the claim that the well-being of humanity, as a whole, has been continuously increasing despite a decline in the ecosystem services. Four possible explanations that can be offered in rejection of this claim: 1) human well-being is not measured correctly; 2) human well-being is linked to food production, and not to other ecosystem services; 3) technology has decoupled well-being from ecosystem services; and 4) a time-lag between degradation of ecosystem services and appearance of their effect on well-being; therefore a decline in ecosystem services will affect human well-being in the future (Raudsepp-Hearne et al., 2010).

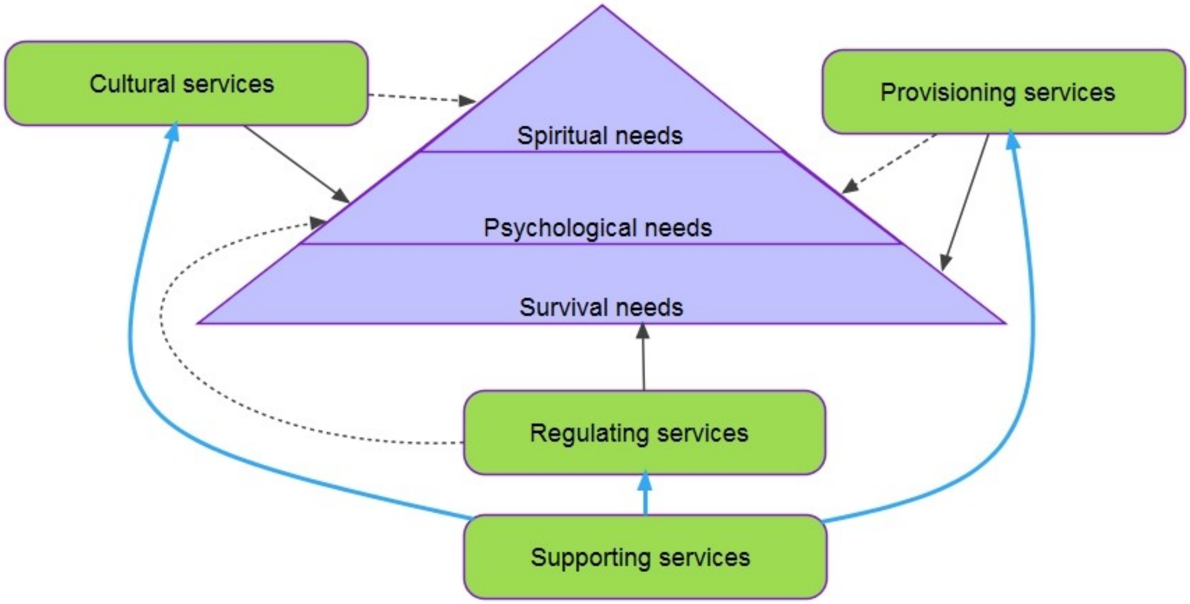

Figure 1.3: The direct (black arrows) and indirect (dotted arrows) contributions of ecosystem services in fulfilling human needs. The supporting services (blue thick arrows) enable other types of ecosystem services. Modified from Geijzendorffer et al. (2017).

1.1.3 Challenges in the field of ecosystem services

One anticipated outcome of the introduction of the concept of ecosystem services was that, by more concretely articulating the benefits of nature for society, this would support and enhance the demands for increased efforts to conserve biodiversity and protect ecosystems (Birkhofer et al., 2015). There has been a fivefold increase in publications with as key term "ecosystem services" in this decade compared to the last one (Olander et al., 2017). The total percentage of published articles containing this key term in the title, keywords, or abstract by country of origin from 2005 to 2016 are: United States, 30%; United Kingdom, 12%; China, 10%; Germany, 9%; Australia, 7%; and Canada, 5% (McDonough et al., 2017). After seminal publications on ecosystem services (e.g., Costanza et al., 1997; Daily, 1997) and the Millennium Ecosystem Assessment report (2005), there has been a significant increase in the number of research papers on ecosystem services, with most of the studies focused on provisioning and regulating, rather than cultural and supporting, categories (Adhikari and Hartemink, 2016). Despite this huge growth in the ecosystem services' literature (Figure 1.4), many challenges remain such as conflicting terminology for ecosystem services, research methods, classification schemes, etc. (McDonough et al., 2017). Additionally, three major challenges in the field of ecosystem services research are: 1) ecosystem services still need to be adapted to account for anthropogenically modified ecosystems, 2) appropriate indicators are not used in the mapping of ecosystem services, and 3) understanding of links between different ecosystem services remains limited/insufficient (Birkhofer et al., 2015).

There are a variety of categorization systems, valuation methods and frameworks for valuation of ecosystem services. This diversity of concepts and the consequent outcomes is a source of confusion that may be one obstacle to the inclusion of ecosystem services into decision making – and environmental management – processes (Burkhard et al., 2014).

1.2 Economic valuation of ecosystem services

If ecosystem services contribute to human well-being, which they do, then they also contribute to human economy as well. The valuation of changes in ecosystem services to show their impact on human well-being is an example of human ingenuity (Cordier et al., 2014; Costanza et al., 2014). One of the key factors inhibiting the decision-making community from including the ecosystem research into the decision-making process is that the literature purely focuses on biophysical and ecological assessments, and fails in linking findings to the outcomes that matter to people. Indeed, there is a pressing need to link biophysical assessments to policy relevant indicators (Olander et al., 2017), such as economic values. The valuation of

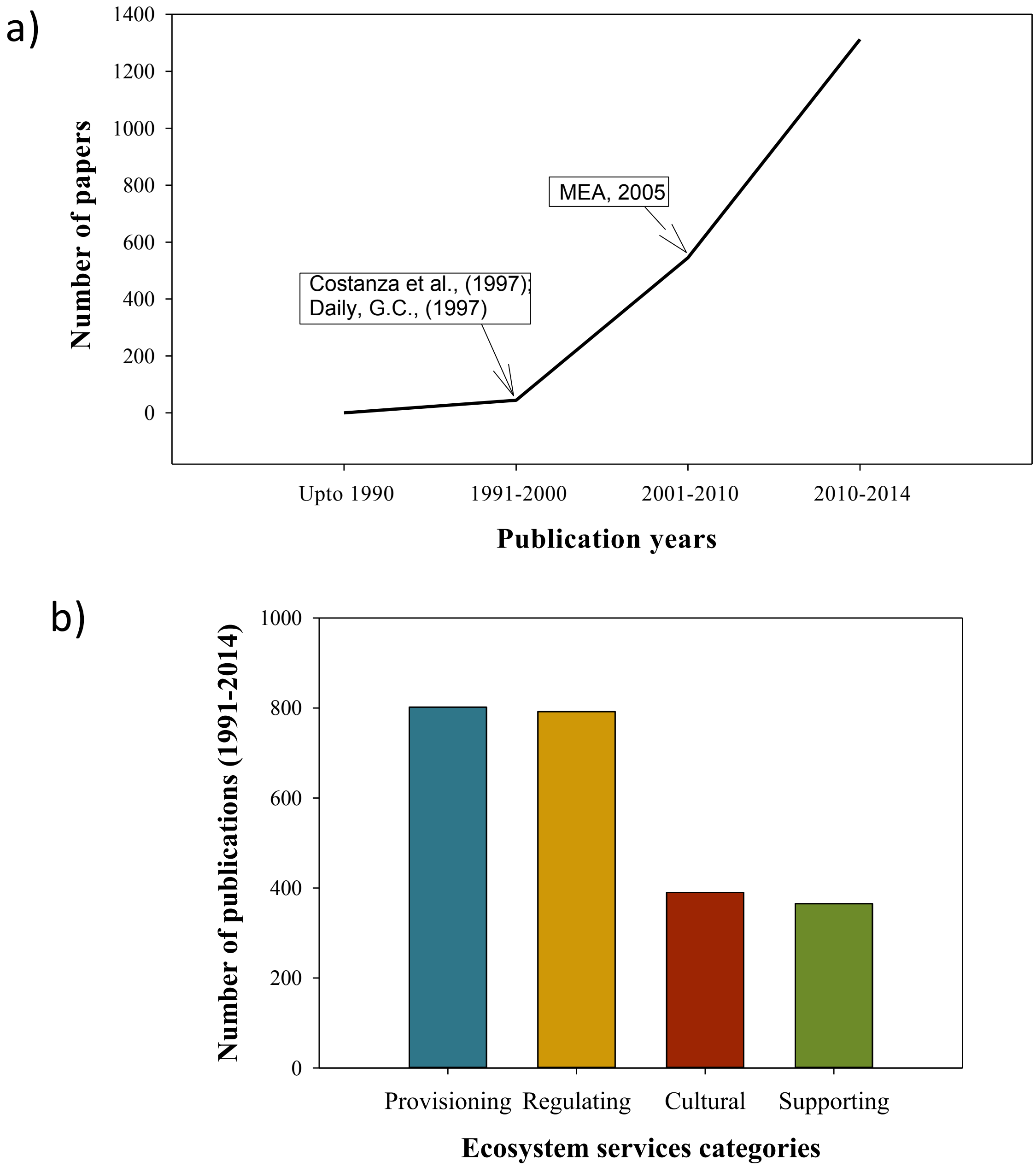

Figure 1.4: Total number of papers published per year a) between years 1990 and 2014 containing the term "Ecosystem Services" in the title. The arrows indicate the publication of influential papers (Costanza et al., 1997; Daily, 1997) and that of the Millennium Ecosystem Assessment (MEA) report (2005) on ecosystem services (Source: Adhikari and Hartemink, 2016), and b) according to the four major types of ecosystem services on average between years 1991 and 2014 (Source: Adhikari and Hartemink, 2016).

ecosystem services is also considered by some as the last and best hope to mainstream conservation efforts. Economic values and taxes reshape human and societal values. In addition, investment for in nature conservation should increase with a realization of its value by institutions and individuals. The appreciation of ecosystems as a valuable stock goes back to Plato or even earlier (Daily et al., 2009), and economists have been valuing ecosystem services produced by natural areas since the 1960s and ’70s (Daily et al., 2009; Krutilla and Fisher, 1975). However, the recent and more concerted efforts aimed at valuation of ecosystem services were greatly invigorated by the powerful vision put forth by the Millennium Ecosystem Assessment in 2005 (MA, 2005). Valuation of ecosystem services has emerged as a new market-based instrument and a dominant global model for environmental management (Jackson and Palmer, 2014; McAfee and Shapiro, 2010; Vatn, 2010). It is important to valuate ecosystem services by linking the biophysical production functions with economics to incorporate them into decision making (Daily et al., 2009), because production functions provide the evidence of ecosystem services as flows into the economy. At the same time, valuation results and their presentation in line with the needs and demands of policy makers can better assist the decision-making process (Wright et al., 2017). On the one hand, valuation addresses the concern that perceived benefits of ecosystem services will be ignored until valuated. On the other hand, the valuation of non-market services has two major limitations: 1) lack of a framework for robust valuation of certain non-market ecosystem services; and 2) lack of necessary information for transferring values from one site to another (Wainger and Mazzotta, 2011).

The payment for ecosystem services (PES) is considered a ‘triple win solution for nature, private investors, and the poor’ in places where marginalized communities are highly dependent on natural resources with government acting on the behalf of beneficiaries (Jackson and Palmer, 2014; McAfee and Shapiro, 2010; Vatn, 2010). This PES model has been implemented in Europe and USA for decades in terms of payments to farmers to adopt soil conservation practices (Gomez-Baggethun and Ruiz-Perez, 2011). While doing the economic valuation of ecosystem services, we need to keep in mind the broader set of goals including social fairness, ecological sustainability and economic efficiency. The human economy has crossed the point where human capital was a limiting factor to a point where natural capital is a limiting factor (Costanza, 2000). It gives rise to a need of better and sustainable management of natural capital. Valuation is not an end in itself, but a tool to help sustainable management of natural resources. It is no panacea but a useful piece of information which can contribute towards the conservation of nature (Costanza, 2006).

1.2.1 Uses of economic valuation

My comprehensive survey of the literature produced an extensive list of the uses of economic valuation of ecosystem services. Valuation can serve (but is not limited to) the following purposes:

- To raise awareness of ecosystem services and their impact on human well-being;
- To avoid ecological scarcity which is increasing due to a lack of appreciation of nature and its flows into the economy;
- To internalize ecosystem services which are usually considered as externalities in the economic system;
- To steer policy- and decision-makers toward sustainable solutions;
- To realize the full economic potential of multiple-user watersheds;
- To more easily identify and quantify different environmental variables in the same units;
- To design compensation schemes for nature;
- To support fair and sustainable resource allocation;
- To select better social choices;
- To inform land use and urban planning;
- To design climate adaptation strategies;
- To understand and capture the impact of human activity on the environment;
- To link assessments to benefit-relevant indicators.

1.2.2 Methods of economic valuation

Ecosystem services are valuated on their direct or indirect uses. Market values are available for ecosystem services that are directly in use in the form of consumptive goods such as food, logged timber, etc. Non-market values are applied for the services that are not in direct use such as preservation value, erosion and flood control. Market values are the direct cost of the product but non-market values are more complicated to derive and often difficult to explain (Kaval, 2010). Non-market values are deduced using different approaches (e.g., stated preference, revealed preference) and production function methods as explained in Table 1.1. The contingent method is the best known direct valuation method for non-market services as compared to indirect methods such as hedonic pricing and travel cost. If data collection required for valuation of an ecosystem service is expensive, difficult and time-consuming, then the benefit transfer method is often used. This method applies values derived for other studied sites using a variety of methods to the site under consideration called the “policy site” (Cordier et al., 2014).

The replacement cost method is the more meaningful and direct way to assess the value of non-market services. In principle, the replacement cost method can be used to valuate non-market values if the "perfect" substitute solution is available, that is, a solution that provides a benefit comparable to that of the original ecosystem. As a caveat, however, it is impossible to value cultural services using the replacement cost method (Ledoux & Turner, 2002). To estimate a meaningful replacement cost requires three conditions to be met: (1) there is a perfect substitute, meaning that the substitute engineered system provides an equivalent function in quality and quantity to ecosystem service, (2) the least cost substitute is chosen, meaning the cheapest human engineered system replaces the ecosystem service, and (3) the public is be willing to incur the cost of the ecosystem service loss. However, a review of replacement cost studies shows that these three conditions are rarely achieved.

Table 1.1: Methods of economic valuation of ecosystem services: description and limitations.

	Method	Description with example[1]	Limitations[2,3]
Non-market valuation methods	The contingent valuation/willingness to pay/ willingness to accept/stated preference	A person states what they will do/pay if a certain situation occurs e.g., money they are willing to pay if they are guaranteed to see ten deer in an area on a particular hiking trip.	Time consuming; biases can occur; sometimes the only method to assess non-use values.
	Revealed preference/ travel cost	People reveal what they did or spent in traveling to a specific site, e.g., money spent on a specific trip to a lake for fishing and camping.	Complicated due to multipurpose trips or trips to multiple places; requires a lot of data; suitable only for direct use.
	Choice experiment/ stated preference	Involves asking respondents a series of questions about different management strategies, e.g., number of picnic areas, percentage of trees that can be harvested.	Hypothetical and biased because based on respondents; requires respondents to weigh the choices.
	Revealed preference/ hedonic pricing	People pay for the environmental resources surrounding the site; e.g., higher prices of houses fronting the beach compared to those blocks away.	Requires a lot of data on actual behavior of people; limited to property benefits.
Sundry methods	Benefit transfer or value transfer	Uses values from contextually similar studied sites applied to the policy (new) site valuation due to time and financial constraints. It can be used for transferring both market and non-market values.	Inaccuracies are generated if extreme care is not exercised in its application; some services valued in more detail than others in existing valuation studies.

Market-based/Indirect market valuation methods	Avoided cost	The cost we do not pay in the presence of the ecosystem service(s): e.g., when a wetland is drained, many ecosystem services including water regulation, water filtration, nutrient cycling, climate regulation, etc. are lost.	Limited to property, assets and economic activities; values can be overestimated.
	Replacement cost	The cost to replace ecosystem services with man-made products, e.g., fertilizer application in the absence of nutrient cycling.	For direct use only; can overestimate; replacement service may only provide a portion of the full range of services.
	Restoration cost	The cost to repair damaged natural ecosystems; e.g., clean-up costs of Exxon Valdez Oil Spill in 1989.	Relates to costs and not to preferences; complete restoration is not possible; restoration takes time.
	Productivity /Factor income/Derived value	Used to capture value of an ecosystem service that enhances the value of commercially marketed goods, e.g., bees pollinate the crop flowers sold in market or ecosystem filtration lowers purifying costs of treating municipal water.	Difficulties in obtaining data for both changes in ecosystem services and their impact on production; relationship between ecosystem service and output product is complicated.

[1](Kaval, 2010); [2](Austin et al., 2012); [3](Spurgeon and Cooper, 2011)

1.2.3 Methodology for economic valuation

Market price is a balanced fulcrum between supply and demand, and it leads to the efficient resource allocation of goods and services. Markets use the feedback mechanism of maximizing profit to adjust prices, which is assumed to encourage human well-being. However, ethical beliefs make the relationship between economy and ecosystems more complex (Daly and Farley, 2010). For example, willingness to pay will be different in the case where a forest is valued for its water filtration function than the case where it is believed to be sacred. At the same time, valuation methods cannot capture all aspects of ecosystem services, such as normative and ethical aspects, and certain conditions such as nearing an ecological threshold or tipping point (Rasul et al., 2011).

Putting a value on non-market services can be a good way to offer insights to public and policy makers, and hence promote appropriate economic policies. However, the use of valuation to justify destroying a given ecosystem is an example of economic imperialism (Daly and Farley, 2010). Generally, the value of non-market ecosystem services is accrued based on land use area and unit value datasets using the following equation (Kreuter et al., 2001):

$$ESV = \sum(A_k \times UV_k) \qquad (1.1)$$

where ESV is the total value of ecosystem services, A_k is the area of land use k, UV_k is the unit value of land use k, and the sum is taken over all land uses in a given region of interest (e.g. a river basin or watershed).

The unit values are generated using biophysical assessments and market values. However, the unit values can be partially or fully local, based on the availability of data. Using non-local datasets, either biophysical properties or market values, will introduce uncertainties in the valuation process (Figure 1.5). Locally developed unit values can reduce transfer errors — errors that arise from the application of value transfer functions — in the value estimates (Schmidt et al., 2016).

1.2.4 Biophysical assessment and local values

The ecological underpinning of economic valuation needs the inclusion of biophysical assessments of ecosystem services in the process (de Groot et al., 2010; Pandeya et al., 2016). These assessments can mainstream the ecosystem services valuations for their incorporation into decision making at local, national and global scale (Pandeya et al., 2016). Similarly, to make value estimates policy-relevant at the local level, there is a need to use local data such as locally monitored rainfall, livestock and agricultural data. Yet, the lack of relevant data at the local scale is frequently a big obstacle to carry out local valuation (Pandeya et al., 2016). On the other hand, people valuate ecosystem services based on their scale-perception, direct benefits and socio-economic background; these values may differ from the ones based on their biophysical assessments (Chaikaew et al., 2017).

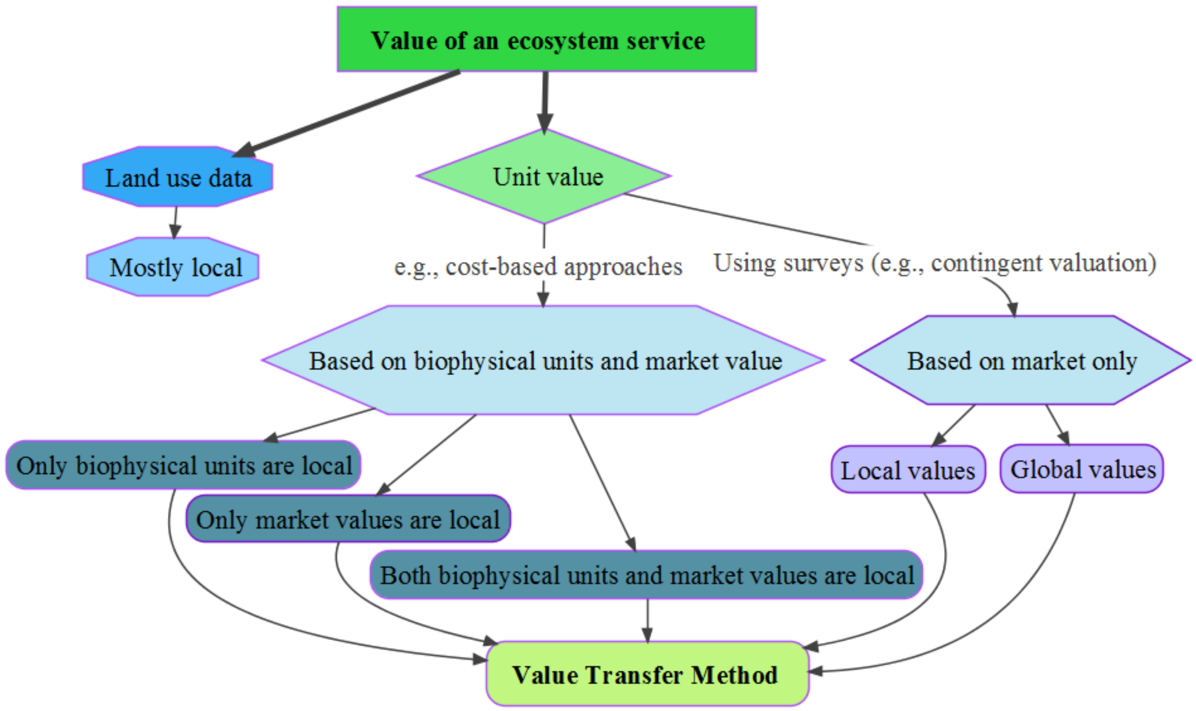

Figure 1.5: Conceptual framework for accruing the value of a non-market ecosystem service using the value transfer method.

1.2.5 Uncertainties in the values of ecosystem services

It is difficult or nearly impossible to quantify ecosystem services without uncertainty because the scientific knowledge of all the relevant environmental, ecological, social and economic processes is incomplete. The science of ecosystem services integrates multiple disciplines (e.g., ecology, hydrology, geospatial, economics) and, therefore, inherits uncertainty from different sources, such as problem context (e.g., future development, future climate, technological options, eco-socio-political context), input data (e.g., land use data, climate data, water treatment cost data, agriculture yields), model structure and model parameters (Hamel and Bryant, 2016). The uncertainties in economic values are broadly classified into three types: supply/biophysical uncertainty (i.e., in the supply of ecosystem services); structural uncertainty (i.e., from the number of ecosystem services selected and the benefits attached); and parametric uncertainty (i.e., valuation methods) (Boithias et al., 2016; Hejnowicz and Rudd, 2017). For these reasons, researchers need to acknowledge and report uncertainties to decision-makers in a systematic and quantified manner. On the other hand, decision-makers should move away from a cookbook approach and incorporate uncertainty in their decisions using strategies (e.g., resilient and adaptive strategies) that will work well for a number of uncertain outcomes (Office of Best Practice Regulation, 2014).

1.3 Anthropogenic drivers of change and appropriate spatial scales for the assessment of ecosystem services

Humans impact ecosystem services by altering ecosystem functions through changes in land use, climate and biogeochemical cycles (Isbell et al., 2017). Anthropogenic activities most readily change the land use resulting in alterations in the flows of ecosystem services (Nelson et al., 2006). In addition, the most pervasive socio-economic driver causing degradation of ecosystems is land use change. At the same time, the selection of the scale of observation establishes the relative coarseness or fineness of details and data considered when assessing ecosystem services. Many ecosystem services (e.g., water quality, drinking water) are produced at regional (watershed) scale, and regional land use plays a key role in their production (Raudsepp-Hearne and Peterson, 2016). Accordingly, at a watershed scale, land use change can be considered an appropriate driver to analyze the subsequent changes in the ecosystem services.

1.3.1 Watershed ecosystem services

A watershed is a region enclosed by watershed lines (Vincent and Soille, 1991) that captures the precipitation and directs/channelizes it to a single outlet draining it into a water body (e.g., lake). Watersheds are based on hydrological boundaries, which make them attractive water systems for scientists to study and for managers to manage. In addition, watersheds are socio-economic units where upstream and downstream users are linked through water flow and the associated ecosystem services. Where watersheds are treated as management units, it becomes even more important for researchers/scientists to asses them as integrated systems (Aguilar-González et al., 2015).

Ecosystems exist within the biophysical context of the watershed and the availability of abiotic factors (e.g., water, soil, air) determines ecosystem types in a region. For example, swamp forests need abundant water throughout the year, fish need freshwater, soil type defines vegetation type, and nutrient supply determines plant growth rates. Alternatively, anthropogenic disturbances also affect the rate of plant growth (i.e., by supplying nutrients via fertilizers) and types of species within an ecosystem (i.e., by planting a specific species in a forest). However, in both cases, abiotic factors underpin the existence and resilience of ecosystems and their services. The dominant drivers that result in land use change (and the associated supply of ecosystem services) within a watershed are socio-economic and political in nature. Thus, the watershed's biophysical context is governed by the human context. For example, decisions of farmers and land use planners impact the land use resulting in changes in abiotic factors and related ecosystem services (Aguilar-González et al., 2015). Therefore, it is imperative to understand the implications of decisions regarding land use in terms of representative indicators of abiotic factors.

1.4 Ecosystem services and land use changes

Land use change substantively affects the suite and magnitude of ecosystem services provided by a given landscape (Dallimer et al., 2015; Tianhong et al., 2010). The ongoing rapid pace of changes in landscape structure makes sustainable management of land use a key challenge in the field of environmental management. A current research priority is to bridge the gap between land use planning, ecosystem services and decision-making process (Schmidt et al., 2017). Often, research focused on valuation of temporal land use scenarios ignores the impact of time lags on the flow of ecosystem services. Studies have shown evidence that many ecosystem services (such as above ground carbon density and recreational usage) are strongly influenced by past land cover ranging from 2 to 100 years ago (Dallimer et al., 2015).

The valuation of ecosystem services can successfully depict the impact of land use change on the abiotic factors that support ecosystem services. For example, land use change impacts on water, soil, and air can be represented by evaluating the water supply, water filtration, nutrient cycling and carbon sequestration services. These ecosystem services can effectively reflect the changes in land use because these changes directly affect the processes underpinning the services as described below.

1.4.1 Water supply

All life on Earth needs water supply in suitable quantity and quality for proper functioning (EPA, 2015). Ecosystems are the part of the water cycle through the processes of evapo(trans)ration, infiltration, runoff and aquifer recharge, whereas the hydrological cycle links different ecosystems and their components (Graymore, 2005). Natural ecosystems such as forests and wetlands slow down water movement on the earth's surface and retain it. These ecosystems recharge groundwater with retained water after filtering out a large portion of the sediments and nutrients. The high quality groundwater supplies multiple direct (e.g., agriculture, livestock, industry, sanitation and drinking) and indirect benefits (e.g., tourism, wildlife, and spirituality) (CGIAR Research Program on Water: Land and Ecosystems (WLE), 2015; EPA, 2015).

Building of large reservoirs and dams has increased the freshwater provisioning services to agroecosystem and other municipal uses giving 2.4 billion people access to water supply in the last 20 years alone. Water withdrawals from inland water systems have increased by 15 times in the past two centuries. Consequently, humans control and have access to more than the half of the total continental flows (Reid et al., 2005). At the same time, increasing anthropogenic use and exploitation of water resources is reducing the capacity of rivers, lakes, wetlands and groundwater aquifers to ensure water supply security. Similarly, land use changes at the watershed scale have important effect on water yield, and consequently, on the water supply service (Geng et al., 2014).

1.4.2 Water filtration

Sediments contribute to physical and chemical pollution in rivers and lakes. High turbidity results in blanketing of gravel bed by fine sediments in spawning rivers and subsequent loss of fish habitat. In addition, phosphorus and metals sorb to fine clay particles and are transported along rivers. Phosphorus is a major limiting nutrient for eutrophication in the freshwater ecosystems and thus important for freshwater quality (Parsons et al., 2017; Scavia et al., 2014). Phosphorus transport measurements in North America and Europe show that approximately 90% of the total phosphorus flux in the rivers is associated with suspended sediments (Ongley, 1996). The control of freshwater eutrophication — richness of nutrients and increased algal biomass — is primarily associated with the control of phosphorus loading in the streams and lakes. For example, increased phosphorus inputs to Lake Erie from 1960 to 1970 degraded the water quality and reduced the central basin's hypolimnetic oxygen levels, which destroyed thermal habitat important to cold-water organisms and damaged an important benthic macroinvertebrate prey species for fish (Scavia et al., 2014).

The strong link between land use and water quality of adjacent water bodies is widely known. Land use directly impact the hydrology of a watershed (Lee et al., 2009). Several studies (e.g., Postel and Thompson, 2005; Warziniack and Morgan, 2016) have demonstrated the impact of forest cover on the water treatment costs in watersheds. That is, a higher the percentage of forest cover in a watershed tends to lower the cost of water treatment for sediment and nutrients. Similarly, vegetation in wetlands plays a significant role in assimilation and storage of sediment and phosphorus. Wetlands remove phosphorus through biological, chemical and physical processes, and retain it for extended periods. This assimilation of phosphorus depends on the vegetation types and their characteristics. Both types of vegetation, floating and emergent, absorb phosphorus from the water, but they differ in its storage (Reddy et al., 1999). Therefore, land use changes will result in fluctuations in the water filtration service within a watershed.

1.4.3 Carbon sequestration

Carbon sequestration is the removal of carbon dioxide, CO_2, from the atmosphere and its deposition into long-term storage primarily in soil organic matter and growing trees and plants. Photosynthesis is the biochemical process through which plants, in the presence of sunlight, convert CO_2 into organic compounds, which make up leaves, roots and stems (Gorte, 2009). This process captures carbon from the atmosphere (fixation or uptake of carbon) and it remains stored in the plants until they die and decompose. For certain plant compounds, such as lignins, decomposition is a slow process, and therefore the production of these compounds represents a long-term carbon storage for hundreds or even thousands of years. In water-logged soils decomposition of organic matter back into CO_2 takes even more time. Different factors affect soil carbon dynamics: soil type, litter type, and the removal or not of organic material from the site of production (Gershenson and Barsimantov, 2010).

Terrestrial ecosystems (e.g., forests, agriculture, and wetlands) sequester carbon, and stock it below ground (in soils and roots) and above ground pools (in trunks, leaves and branches) (Dierkes, 2011). Different terrestrial ecosystems i.e., (land uses) have different carbon sequestration capacities and carbon stocks in and above the soil (Olewiler, 2004; Tao et al., 2015). For example, conventional tillage crop can sequester carbon at a rate of 0.4 ton per hectare per year (t/ha/year), whereas conservation cover can sequester at much higher rates, up to 1.8 t/ha/year (Olewiler, 2004). Therefore, land use change is a key factor that affects the carbon sequestration potential and terrestrial carbon stocks in regional ecosystems (Tao et al., 2015).

1.4.4 Nutrient cycling

Nutrient cycling is the movement of nutrients within and between the physical environment and living organisms (Lavelle et al., 2005). Efficient nutrient cycling maintains the fertility of the soil and helps plant growth. Sixteen chemical elements important for plant growth are obtained from soil and air. Carbon (C), oxygen (O), and hydrogen (H) are obtained from air and water and, therefore, usually not considered nutrients. These three elements are used up in making the bulk mass of plants through photosynthesis. By contrast, nitrogen (N), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), sulfur (S), iron (Fe), manganese (Mn), zinc (Zn), copper (Cu), boron (B), molybdenum (Mo), and chlorine (Cl) are obtained mostly from the soil and are needed by all plants for their growth. All of these nutrients are equally important for plants, but differ in the amounts required: N, P and K are the primary macronutrients and are required in large amounts, whereas Ca, Mg and S as secondary macronutrients are required in smaller amounts; Fe, Mn, Zn, Cu, B, Mo, and Cl are micronutrients required in the smallest amounts (Bierman and Rosen, 2015).

Although all ecosystem services are supported by a balanced supply of nutrients through the nutrient cycling processes, human activities disturb this balance. The potential of soil, sediment, or water to supply nutrients is called fertility, which is a supporting service for many other provisioning services such as food, fiber, fuel and timber supply. Fertility also supports other ecological processes that are vital for ecosystem stability. However, human activities, mainly associated with agriculture (e.g., use of synthetic nitrogen fertilizers, mining of P), have altered the cycling of the key nutrients (N, P, K) over the past two centuries. The human-managed systems (e.g., agriculture) are supplemented through fertilizer application to sustain or enhance soil fertility. Meanwhile, excessive supply of nutrients from these systems causes eutrophication in the receiving water bodies (Lavelle et al., 2005). Land use conversion (e.g., from natural to agricultural and urban land covers) caused by human activities and the resulting erosion processes reduce soil fertility (Hartemink, 2010).

1.4.5 Importance of land use change scenarios for ecosystem services

Based on the selection of scenarios, the valuation of ecosystem services can be broadly categorized into two types: sustainability analysis and program evaluation analysis. First, sustainability analyses assess how changes occurring today due to economic development impact the sustainability of natural ecosystems that underpin economic activity. Second, program evaluation analyses provide comparative assessments of future development strategies. Therefore, program evaluation analyses involve one or more scenarios under the influence of different drivers such as environmental change, local and global market trends, and policy shifts (Bateman et al., 2011).

1.5 Unit values of ecosystem services for different ecosystem types

The first step in developing an economic valuation framework involves recognition of how different types of ecosystems deliver a given ecosystem function or service. The capacity of an ecosystem to generate ecosystem services depends on its functions (de Groot et al., 2002), which, in turn, are dependent on ecosystem type (e.g., forest, wetland, agroecosystem). Most watersheds are multifunctional landscapes and contain a mosaic of ecosystems. All functioning ecosystems provide ecosystem services to some extent; however, the magnitude of services depends on the ecosystem type. At the same time, it is now widely recognized that natural ecosystems provide more services (in number and magnitude) than agricultural ecosystems (Felipe-Lucia and Comin, 2015). Accordingly, the values of different ecosystems differ in terms of their ecosystem services (Figure 1.6).

In the economic valuation literature, ecosystems with varying functions are sometimes aggregated under a single land use category for the purpose of valuing their services. For instance, wetlands are one of the most valuable, but also very diverse, ecosystem class that has many types. Therefore, there is a need to introduce frameworks that capture and distinguish the values of these different types of wetlands.

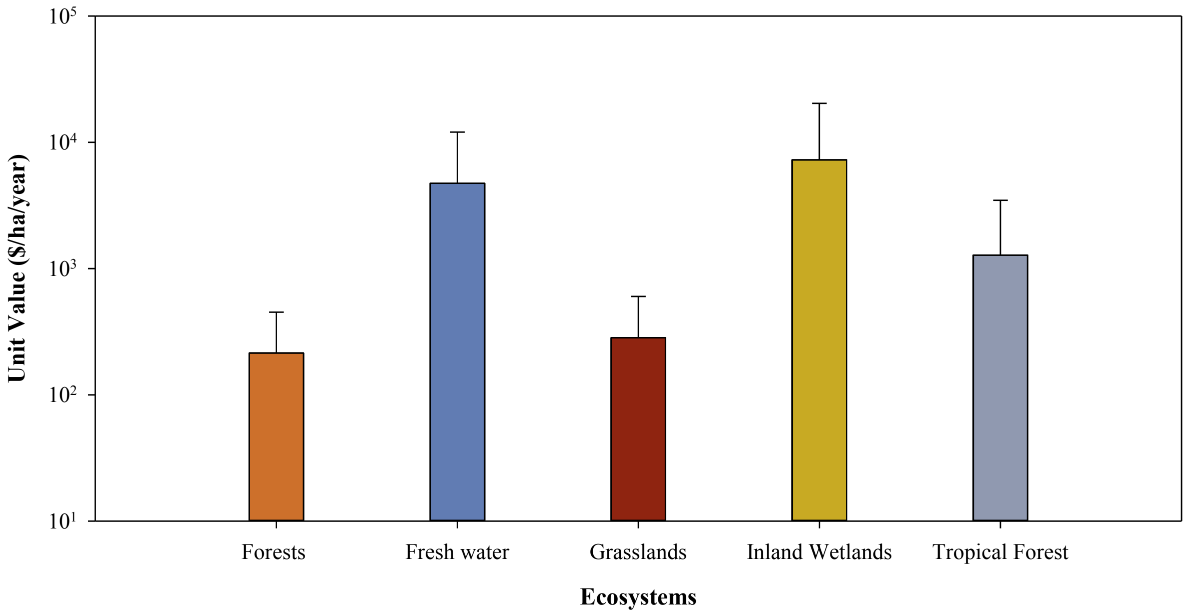

Figure 1.6: Average unit values (CAD 2017) of ecosystems worldwide. The "Forests" category represents both temperate and boreal forests. (Source: de Groot et al., 2012). The error bars represent standard deviations in the mean unit values. Note the logarithmic scale.

1.6 Unit value datasets

The value transfer method is time- and resource-efficient in conducting economic valuation of ecosystem services. However, it uses unit values ($/hectare/year) from the primary studies, that is, first-hand monetary appraisals of ecosystem services conducted at other sites. McVittie and Hussain (2013) showed that 23% of total value estimation studies are based on the value transfer method, making it the second largest after the direct pricing method which is used in 37% of the studies. There are significant variations in unit values due to availability of a large number of primary studies worldwide (Schmidt et al., 2016). There exist global datasets of unit values that cherry-pick values from the primary studies conducted across the globe, and these datasets are widely being used for valuation estimates. Therefore, there is an urgent need to investigate the transferability of these datasets for use in regional- and local-scale valuation studies.

1.7 Value of the used portion of ecosystem services

Most of the provisioning services are easily quantifiable based on their actual use by people. Yet, the quantification of regulating, cultural and supporting services based on their use is a complicated process. For example, it is easy to determine the number of people depending on a water supply from the watershed; but to locate the users of the carbon sequestration service provided by a forest stand is an ambiguous task because it is a global service. The value of realized ecosystem services is directly proportional to the population density of a region; therefore, it will vary from region to region (Turner et al., 2012). Because realized ecosystem services are actually used by the people, their value matters more to the users and policy-makers.

1.8 Research questions

This thesis addresses the need to make meaningful economic estimates of ecosystem services that can inform policy-making. To achieve these objectives, I formulate overall research questions (RQ), which are further broken down into sub-questions (SQ):

RQ 1: What is the impact of long-term land use changes on the three key abiotic factors (air, water, and soil) in terms of indicative ecosystem services in a large, multi-use watershed (here the Grand River watershed in Ontario, Canada)?

SQ 1.1: How do we account for the value of ecosystem services in anthropogenically modified watersheds?

SQ 1.2: What is the difference in the value of consumptive and non-consumptive ecosystem services in response to land use change?

SQ 1.3: Do local unit values reduce uncertainty in the valuation estimates?

RQ 2: How do we value wetland types based on the magnitude and level of services they provide?

SQ 2.1: What is the value of water filtration provided by four major wetland types in southern Ontario?

SQ 2.2: How much will it cost to replace the water-filtration service provided by all wetlands in southern Ontario with human-made infrastructure?

RQ 3: What unit-value and land use dataset can improve the application of the value transfer method

SQ 3.1: How reliable are the global unit-value datasets compared to local and regional datasets?

SQ 3.2: How does resolution of the land use data affect the predicted values of ecosystem services?

RQ 4: How do we distinguish between values of used and non-used portions of ecosystem services?

SQ 4.1: How much of the potential ecosystem services are realized in southern Ontario?

SQ 4.2: What type of ecosystem services maps can better help guide investment in natural infrastructure to maximize their societal benefits?

1.9 Thesis Structure

In this thesis, I address the four major research questions (RQ1-RQ4) on the economic valuation of ecosystem services. Therefore, my research work comprises four chapters (Chapters 2-5):

Chapter 2: I valuate four ecosystem services, including a market and three non-market services, in the Grand River watershed located in southern Ontario, Canada. The analysis shows variations in the ecosystem services values for four temporally explicit land-use scenarios. In this work, I address the challenge of taking into account ecosystem services from anthropogenically modified systems. My results show the importance of using local unit values when valuating ecosystem services. Further, I demonstrate that value of consumptive and non-consumptive ecosystem services depend on different drivers and hence they need different economic approaches for their valuation.

Chapter 3: Wetland types differ in their capacity to generate and supply ecosystem services because of their distinct features. Therefore, I develop a framework based on wetland types to assess the value of ecosystem services. Next, I apply the framework to four major wetland types in southern Ontario, Canada, and show difference in their water filtration service values.

Chapter 4: Many studies have used global datasets and the value transfer method to value ecosystem services in different parts of the world. The value transfer method is quick but gives a crude approximation due to its reliance on a variety of methods and datasets. My work highlights the wide differences and inconsistencies in the output of a valuation exercise using different datasets of unit values and land use.

Chapter 5: In this chapter, I distinguish the realized ecosystem services from the potential ecosystem services and value them. Because of their direct contribution to human well-being, the value of realized ecosystem services matters more to decision-makers than the potential ecosystem services. I develop a map for realized ecosystem services in the study region to locate hotspots for potential investments in natural infrastructure.

Chapter 2

Valuation of four ecosystem services in response to land use changes in the Grand River watershed, Ontario, Canada

2.1 Summary

Economic valuation of ecosystem services based on local or regional data provides dependable information on the economic implications of land use changes that can support decision-making in watershed management. This paper presents a locally-based valuation of four ecosystem services in the Grand River watershed located in southern Ontario, Canada. The watershed has a drainage area of 6800 km^2 of which about 80% is under agriculture. The watershed has undergone profound land use changes since European settlers first arrived, and these will likely continue into the future. To illustrate the impact of evolving land use, we select four ecosystem services: water supply, water filtration, carbon sequestration, and nutrient cycling. Water supply is a consumptive service valuated based on water usage in the watershed and the wholesale drinking water rate. The other three non-consumptive ecosystem services are valuated by deriving local unit values that account for the local context by linking biophysical assessments of ecosystem services with local market values using cost-based methods. Four land use scenarios are considered, the present-day situation (Year 2015), and three hypothetical scenarios. The latter approximate the land use and demographics in the watershed prior to European settlement (Year 1800), in the midst of the Second Industrial Revolution (Year 1900), and in the middle of this century (Year 2050). These scenarios illustrate the impacts of agricultural intensification, urbanization and watershed management on the supply and value of ecosystem services. The combined monetary value of the four ecosystem services is maximum for the pre-European settlement scenario due to the high percentages of forest (84%) and wetland cover (16%). The least value is obtained for the year 1900 scenario due to the decreases in forest (down to 5%) and wetland covers (down to 10%) caused by the expansion of agriculture. The current land use scenario shows an increase in value because of the recovery of forest area to about 10%. The target land use scenario for 2050, which calls for a forest cover of 30%, shows further increase in value of the selected ecosystem services in the watershed. The valuation framework presented here can readily be extended to other ecosystem services. By conducting the economic valuation of ecosystem services along a hypothetical, but evidence-based, land use trajectory it provides a broader context in which to make land use planning decisions. Our analysis also emphasizes the different drivers of the values of consumptive and non-consumptive ecosystem services and, hence, the need for different approaches to their valuation.

2.2 Introduction

Ecosystem services are the direct or indirect benefits obtained from ecological systems (Troy and Bagstad, 2010), and natural capital is the stock of natural resources yielding these services (Olewiler, 2004). Ecosystem services are at the core of the premise that improving the health of ecosystems ultimately improves human well-being (Bouma and van Beukering, 2015). Ecosystems support both consumptive and non-consumptive uses. Consumptive use involves the physical extraction of a component from an ecosystem for human usage, e.g., fish and timber harvesting, whereas non-consumptive uses do not, e.g., recreational services and pollination (Heal et al., 2005). Overexploitation, largely as a consequence of extraction or pollution, of ecosystems has resulted in the degradation of ecosystem services. Only a change in current policies and practices can reverse further ecological degradation (Daily, 1997; Lara et al., 2009; Reid et al., 2005).

Watersheds are hydrological units with interlinked biotic and abiotic components (Berkes et al., 1998). Watershed divides act as ecological boundaries (Bormann and Likens, 1979). Conservation authorities in southern Ontario, Canada, have been given the mandate to protect and manage natural water and land resources at the watershed scale. Therefore, watersheds are also operational management units and the Province has been recognized as a world leader in watershed management (Conservation Ontario, 2003). River watersheds provide a myriad of ecosystem services, including supporting, regulating, provisioning and cultural services (Reid et al., 2005) Smith et al., 2006).

Economic valuation of ecosystem services is the allocation of monetary values to ecosystem services. However, it is near-impossible to capture the complete depth and breadth of ecosystem services (Asah et al., 2014). Ecosystem services may have market or non-market values. A direct market value is the cost of a product in the market. These values are most often available for ecosystem services that are directly used in the form of consumptive goods (e.g., water supply, food, logged timber, etc.) (Kaval, 2010). Non-market values are assessed when there is no existing market involving money for the given ecosystem service (Freeman et al., 2014; Hartwick and Olewiler, 1986; Kaval, 2010; National Research Council, 2005; Pearce and Turner, 1990; Tietenberg and Lynne, 2012) . Non-market or indirect market values are most often applied to non-consumptive services (such as water filtration, carbon sequestration, nutrient cycling, preservation value, erosion and flood control) which are more difficult to valuate, and more difficult to advocate for, due to the absence of markets (Kaval, 2010). The production of market services often relies on the non-market services, and both of these (market and non-market) services depend upon the land use and management intensity (Ghaley et al., 2013). Non-market services need indirect valuation approaches which may require measuring the value of changes in ecosystem services.

Most previous watershed valuation studies of non-market services in the literature (Table 2.1) apply the approximate method of benefit transfer at national and global scales (Barton and Mourato, 2003; Brouwer and Bateman, 2005; Chotikapanich and Griffiths, 1998; French and Hitzhusen, 1999; Yongguan et al.,

2001) where existing estimates from similar, already studied, sites are used to valuate new or "policy sites" (Johnston et al., 2015). Valuation studies conducted at different geographical sites and for different socio-economic conditions produce different results (Brouwer, 2000). Even the same type of ecosystems at different geographical locations yield different values due to differences in their ecology and ecosystem functions.

The supply of ecosystem services is mainly altered by land use change, as it affects ecosystem structure and functions (Fürst et al., 2013; Palomo et al., 2014; Si et al., 2014; Su and Fu, 2013; Vitousek et al., 1997; Yang et al., 2009). The supply of ecosystem services also depends on how an ecosystem interacts with the atmosphere, aquatic systems and the surrounding land (Vitousek et al., 1997; Yanai and Lucash, 2003). An understanding of the relationship between ecosystem services and land use change is necessary to maintain a sustainable flow of ecosystem services (Fang et al., 2014). Valuation is a feedback mechanism (Brondízio et al., 2010) which can be used to assess land use change decisions with regard to ecosystem services. Land use changes can lead to synergies (increase in one service increases another) or tradeoffs (increase in one service decreases another) between ecosystem services (Smith et al., 2013). Therefore, valuation of land use scenarios can help better manage the ecosystem services in a watershed.

In the valuation studies reviewed in Table 2.1, all of which are based on the benefit transfer method, many ecosystem services are not valuated because of a lack of relevant data. For example, in the case of the Mackenzie valley watershed in Canada (Anielski and Wilson, 2010) and the Nisqually watershed in the United States (Schmidt et al., 2011), nutrient cycling is not valued due to the absence of adequate primary studies and unit values. The authors of these studies explicitly state that a key limitation is the use of the benefit transfer method. A valuation approach based on biophysical units of ecosystem services is believed to be more accurate due to its dependency on the changes in ecosystem processes (e.g., biomass production).

Braat and ten Brink (2010) describe a functional relationship between land use intensity, biodiversity and ecosystem services, wherein the increase in the land use intensity decreases the flow of net ecosystem services. Human domination of biosphere has led to the decline of ecosystem services due to alterations in the structure and functioning of ecosystems (Kremen, 2005; Vitousek et al., 1997). Ecosystem services are dependent on the availability of stocks of natural resources, such as fertile soil, clean water and stable atmospheric composition (Cork et al., 2007). These abiotic resources can be depleted or deteriorated on a human time scale (1-50 years), while being renewable on geological time scales (van der Meulen et al., 2016). Therefore, the effects of land use change and abiotic factors on ecosystem functions is as, or even more, important than biodiversity (Alcamo et al., 2005). Geo-physical processes create the physical template of a watershed and are dominant in its functioning (Zalewski and Naiman, 1985). At the same time, biotic processes and disturbance regimes interact within this template to generate ecosystem services (Zalewski and Naiman, 1985). Therefore, we illustrate the impact of land use changes on the three core resource stocks (water, soil and air) using four representative ecosystem services in a watershed.

An increasing trend in the gray literature on ecosystem services valuation in watersheds is the transfer of unit values from primary studies that are conducted at different scales and in different environmental settings. Most of these studies (see Table 2.1) have loopholes in their economic analysis, in particular by leaving out several land use categories due to the absence of primary studies. Similarly, Costanza et al. (1997) in their landmark study on valuation of ecosystem services did not valuate many of the ecosystem services provided by agricultural land cover such as, for example, carbon sequestration, nutrient cycling, and water flow regulation. By contrast, economic valuation studies in the peer reviewed literature carried out at the watershed scale are relatively scarce.

In the present study, we generate local unit values for four selected ecosystem services in the Grand River watershed, Ontario. Because the unit values reflect the local context in the watershed they are more directly relevant for decision making purposes than unit values retrieved from global data sets (Pandeya et al., 2016).

Table 2.1: Ecosystem services valuation studies conducted at watershed scale.

Name of watershed and location	Nature of watershed	Team and year of study	No. of eco-services valued	Total Area (10^3 hectares)	Value/year (CAD $)	Unit Values ($/ha/year)
Skykomish Watershed, Washington State, USA	Mostly forested (53%), shrubs (22%)	Earth Economics Team, 2011	17	217	245 million to 3.3 billion	1,130 to 15,230
Lake Simcoe Watershed, Ontario, Canada	Mostly agriculture (40%) and forest (20%)	The David Suzuki Foundation, 2008	13	331	975.2 million	2,946
The Middle Cedar River Watershed, Iowa, USA	Mostly agriculture (86%), forest (2.5%)	Earth Economics Team, 2012	14	604	550 million to 1.9 billion	910 to 3,145
McKenzie Watershed, western Oregon, USA	Mostly forested (72%)	Earth Economics Team, 2012	16	344	247 million to 2.4 billion	720 to 6,830
British Columbia's lower mainland, Canada	Forest (40%), Marine (42%)	The David Suzuki Foundation, 2012	10	2,975	30 billion to 61 billion	10,085 to 20,500
The Puyallup River Watershed, Washington State, USA	Mostly forested (54%), shrubs (11%)	Earth Economics Team, 2011	18	273	526 million to 5 billion	1,925 to 18,320
Snoqualmie Watershed, Washington State, USA	Mostly forested (70%)	Earth Economics Team, 2010	13	180	257 million to 2.4 billion	2,930 to 13,515
Nisqually Watershed, Washington state, USA	Mostly forested (67%)	Earth Economics Team, 2009	12	178	165 million to 3.3 billion	925 to 18,490
Mackenzie Valley Watershed	Mostly forested (50%)	Canadian Boreal Initiative, 2010	17	166 x10^3	570.6 billion	3,425
Lake Winnipeg watershed	Mostly agriculture (56.6%)	International Institute for Sustainable Development (IISD), 2008 (Voora and Venema, 2008)	7	5,670	0.3 to 1.3 billion	60 to 230
Barnegat Bay watershed, New Jersey, USA	Mostly forested (36%) and urbanized (22%)	(Kauffman, 2012) University of Delaware, 2012	14	174	1.4 billion to 6.1 billion	8,065 to 34,850
Credit River Watershed, Ontario, Canada	Mostly agricultural (33%) and developed (33%)	Kennedy and Wilson, 2009	10	95	371.1 million	3,910
Peace River watershed, British Columbia, Canada	Mostly forested (64%), wetlands (9%) and grassland (7%)	David Suzuki Foundation., 2014	14	5,612	7.6 billion to 9.2 billion	1,355 to 1,630
Snohomish Watershed, USA	Mostly forested (75%), Grasslands (12%)	Earth Economics Team, 2010	15	485	429 million to 5.83 billion	885 to 12,025

All values are converted to 2015 CAD using the inflation calculator from Bank of Canada (http://www.bankofcanada.ca/)

The ecosystem services selected include both market (consumptive) and non-market (non-consumptive) services: water filtration, water supply, CO_2 sequestration and nutrient cycling. Unit values are estimated for the all the major land use classes for which areal data are publicly available.

The non-market services are valued using biophysical assessments (e.g., tons of CO_2 sequestered by tree species, kg of nutrient uptake by forest vegetation or crops) and their values in parallel proxy markets (e.g., water filtration plants, storm water management facilities, cap-and-trade system, prices of commercial fertilizers). To the extent possible, the biophysical units and proxy market values used in this study are based on local plant species and markets. The analysis is further contextualized by performing the valuation of the ecosystem services under four different land use scenarios that reflect conditions representative of the pre-development stage of the watershed, the present time and the foreseeable future: pre-European settlement (Year 1800), Industrial Revolution (Year 1900), present (Year 2015), and mid-century (Year 2050).

We estimate the delivery of ecosystem services for the different land use scenarios taking into account changes in agricultural productivity resulting from agricultural intensification. Valuations are all performed with the current demand per capita (static economic analysis) and market prices, in order to isolate the effect of land use changes. In other words, impacts of hypothetical land use changes are assessed while keeping the ecosystem services' marginal values constant at the current level of provision (for an alternative approach, see Dupras et al., 2016). We further carry out a partial sensitivity analysis on land use changes and ecological scarcity of natural land use categories across the land use scenarios.

2.3 Material and methods

2.3.1 Study area

The Grand River watershed is the largest watershed in southern Ontario, Canada, with a total area of 6800 km^2; it currently has a total population close to one million (Shifflett, 2014). There are more than 100 municipal wells and four river intakes for municipal water supply across the watershed (Shifflett, 2014). The Grand River flows through countryside and urban areas for more than 300 km and drains into Lake Erie. Currently, major anthropogenic land uses in the watershed are agriculture (66%) and urban (5%). Since year 1800, land use has significantly changed from mainly forest cover to mainly agricultural use. Urban sprawl, intensive agriculture and excessive fertilizer application are major threats to ecosystem services in the watershed (Olewiler, 2004).

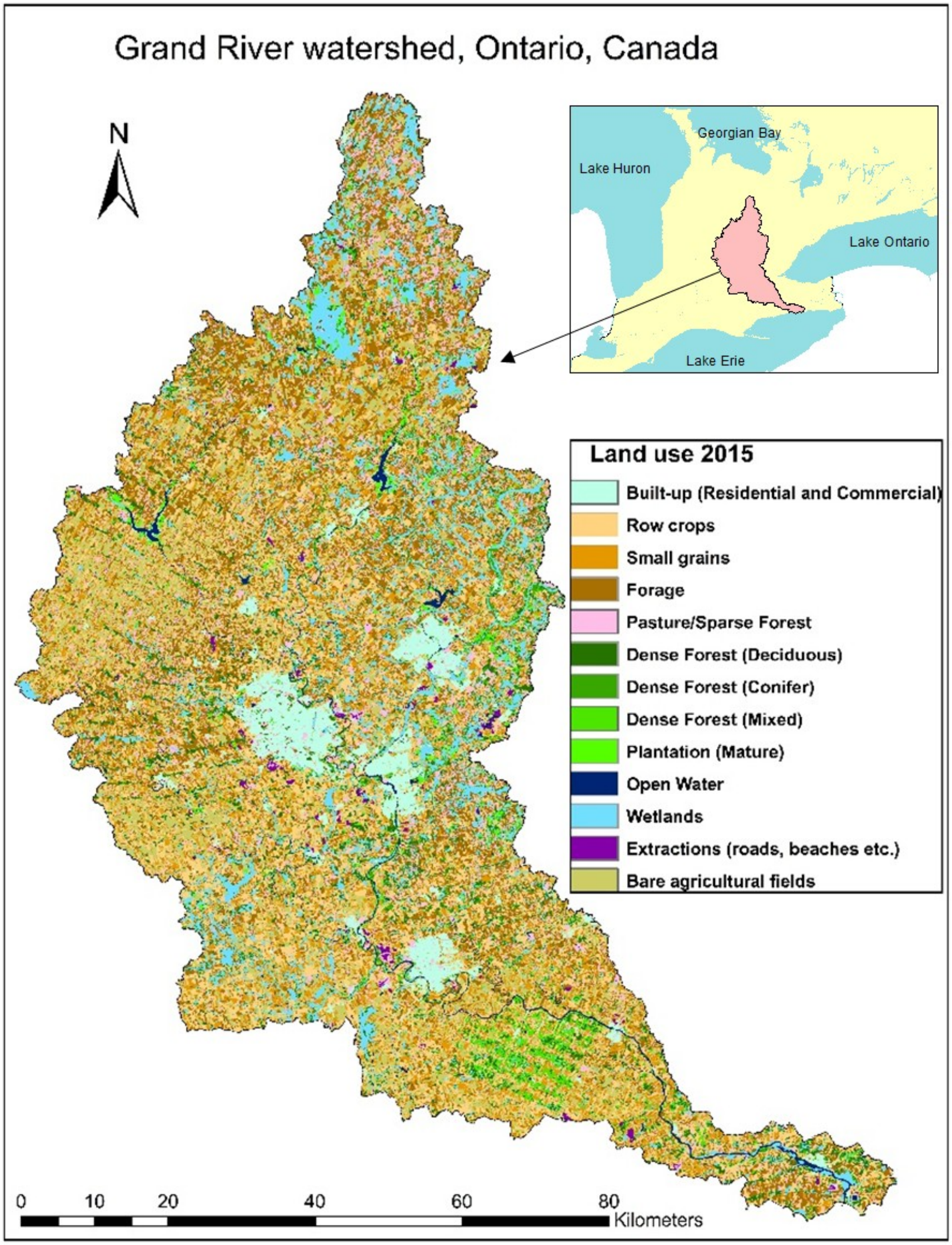

Figure 2.1: Current (2015) land use in the Grand River watershed (GRCA, 2016).

2.3.2 Selected ecosystem services

Terrestrial ecosystem processes regulate water, air and soil quality and provide ecosystem services (Reid et al., 2005). Changes in these processes affect the delivery of ecosystem services and the major driver of these changes is land use (Smith et al., 2013). To assess the impacts of land use intensification and restoration in the watershed on three critical natural resource stocks – water, air and soil – we select the following four ecosystem services:

- Water supply: assess stress on water quantity
- Water filtration: assess impact on water quality
- CO_2 sequestration: assess changes in atmospheric CO_2 regulation
- Nutrients cycling: assess soil capacity to retain and supply essential nutrient elements

The consumptive service is water supply which has a readily available market value. The other three ecosystem services are non-consumptive with no direct market values available.

2.3.3 Valuation method for consumptive service (Water supply)

Groundwater and surface water resources in the watershed are regulated by recharge, storage and discharge processes that operate at the landscape scale. Water supply is a consumptive service and its value depends on the use or rate of resource exploitation. Water supply typically increases with an increase in population. Major water uses in the Grand River watershed are municipal, agricultural, domestic and industrial. The Grand River watershed supplies all of the water needed by its residents; for municipal uses, the supply comes 69% from ground water and 31% from surface water sources (Shifflett, 2014). Water supply has a market value – the bulk water price. Thus, the value of water supply is determined by the market method based on the total water consumption and a wholesale rate for bulk water supply within the watershed (Equation 1):

$$V_w = Q_w * U_w \qquad (2.1)$$

where V_w is total value of water supply ($/year); Q_w is the total water consumption in the watershed (m3/year) and U_w is the unit whole sale rate for water supply in the watershed ($/m3). According to the price elasticity of demand, the demand remains constant for a fixed price (Daw et al., 2016). Therefore, current price and demand for water consumption for municipal (per capita), agricultural and other uses (m^3/ha/year) are applied to the various land use scenarios. For example, unit water consumption for agriculture is calculated dividing the total water consumption of the agricultural sector by the total area under agriculture in the watershed. A similar approach is adopted for the other land uses (Table 2.3).

2.3.4 Valuation method for non-consumptive services

The valuation of non-consumptive services integrates biophysical (ecosystem-based) and economic data. Because the three non-consumptive services do not have direct market values, cost-based methods are used as the alternative approach, in particular replacement or substitute cost and cost avoidance methods. A major advantage of cost-based approaches is that they need less data and resources as compared to methods based on assessing preferences (Notaro and Paletto, 2012). For valuation of these services, we use a variation of the model of Costanza et al. (1997) in which for a given ecosystem service distinct unit values are assigned to each land use category. The unit values integrate the biophysical processes performed by the ecosystems with proxy market transactions and operating costs of engineered solutions, i.e. using cost avoidance and replacement cost methods. Here the unit values are expressed as a mean value and a standard deviation (±SD). The standard deviation is based on variations in both the biophysical variables and the market proxies.

The following four steps are taken to valuate the non-marketable ecosystem services.

1. Identify the land use categories and estimate their surface areas in the watershed.
2. Generate unit values, U_i (in dollars per hectare), for the three non-consumptive ecosystem services in each of the land use categories.
3. Calculate the economic values of the ecosystem services value for each land use category by multiplying the sum of unit values of the selected services generated by a given land use, V_j, by its surface area, A_j.
4. Sum the ecosystem values of the different land use categories in the watershed to obtain the total value TV.

The mathematical formulation of the four-step methodology is given by equations 2.2 and 2.3.

$$TV = \sum_{j=1}^{n} (A_j V_j) \qquad (2.2)$$

where TV is the total value of the selected ecosystem services in the entire watershed ($/year), n is the number of land use categories, A_j is the area of *j*th land use category (ha), and V_j is the sum of the unit values of each ecosystem service provided by the *j*th land use ($/ha/year). V_j is defined by:

$$V_j = \sum_{i=1}^{m} U_i \tag{2.3}$$

where m is the number of ecosystem services, and U_i is the unit value of the *i*th ecosystem service in the *j*th land use category ($/ha/year).

2.3.4.1 Water filtration

Natural land covers such as forests and wetlands act as filters in the watershed (Ernst, 2004). They improve soil stability and resistance to erosion (Belcher et al., 2001; Elmore et al., 2015; Wall et al., 2002). These ecosystems reduce sediment and nutrient loading to surface waters, which in turn reduces water treatment costs by increasing the quality of intake water (Belcher et al., 2001). The intensity of erosion is strongly correlated with land use; vegetation is believed to be the most effective land cover against erosion (García-Ruiz, 2010; Kosmas et al., 1997; Pacheco et al., 2014; Syahli, 2015). In contrast, soil erosion and the associated delivery of phosphorus (P) from intensive agricultural areas are major water quality concerns for water treatment costs, fishing and recreation (Belcher et al., 2001). Phosphorus is of particular importance, as it is the primary limiting macronutrient responsible for algal blooms in the lower Great Lakes.

We valuate the water filtration service for sediment and P removal in the Grand River watershed. Sediment and phosphorus delivery rates are highly variable depending on soil type, tillage practices and slope (Belcher et al., 2001). Therefore, local average rates are used in this valuation study. The unit value ($/ha/year) of the water filtration service (U_{wi}) for each land use is the sum of sediment and P reduction (Equation 2.4):

$$U_{wi} = U_{SF_i} + U_{PF_i} \tag{2.4}$$

where U_{SF_i} is the unit value of relative sediment reduction ($/year) and $\mathrm{U_{PF_i}}$ is the unit value of relative phosphorus reduction for the *i*th land cover ($/ha/year). The sediment delivery rates (sediment yield per unit area, Table 2.4) for different land use categories are obtained from studies conducted within the Grand River watershed and in neighboring watersheds, and verified against regional and global delivery rates (Bender et al., 1997; López et al., 1998; Pimentel and Burgess, 2013; Shaver et al., 1994). The variations in the average sediment delivery rates are expressed as standard deviations in the reported rates by the different studies for each land use category in the watershed.

Equations 2.5 and 2.6 are used to calculate the unit values (\$/ha/year) of sediment filtration (U_{SF_i}) and phosphorus filtration ($\mathrm{U}_{\mathrm{PF_i}}$) by each land use:

$$U_{SF_i} = (SD_{max} - SD_i) * SR_c \qquad (2.5)$$

where SD_{max} is the delivery rate of the land use that delivers the maximum sediment yield in the watershed (tonne/ha/year) or a corresponding regional maximum; SD_i is the sediment delivery rate of the *i*th land use (tonne/ha/year) and SR_c is the sediment removal cost (\$/tonne). For P filtration we use:

$$U_{PF_i} = (PD_{max} - PD_i) * PR_c \qquad (2.6)$$

where PD_{max} is the delivery rate of the land use that delivers the maximum phosphorus in the watershed (Kg/ha/year) or a corresponding regional maximum; PD_i is the phosphorus delivery rate of the ith land use (Kg/ha/year) and PR_c is the phosphorus removal cost (\$/Kg). Data compiled from various literature sources is presented in Table 2.4.

The cost to remove sediment from storm water management facilities (SWMFs) is used as a proxy for SR_c. This proxy is chosen because SWMFs represent the least expensive alternative for sediment removal. The unit rate for P removal (PRc) is the average of 12 Water Pollution Control Plants (WPCPs), a wastewater treatment center (WWTC), and a sewage treatment plant (STP), all located in Ontario. These facilities, which discharge treated effluent into Lake Simcoe watershed, are selected because operating costs for P removal are reported separately from all other functions (XCG Consultants and Ltd., 2010).

2.3.4.2 Carbon sequestration

Natural ecosystems, in particular forests and wetlands, are considered sinks for CO_2 that can help counterbalance emissions of greenhouse gases (Gale et al., 2009). The carbon sequestration rates of different plant species depend on biotic properties (e.g., stand density, insect abundance, mycorrhizae or fungus, tree disease and pathogens) and abiotic factors (e.g., precipitation, soil texture, nutrient availability) (Gale et al., 2009). Regional and global carbon balances are regulated by forest ecosystems as they store 76 % of all carbon in the terrestrial biosphere (Dixon, R.K., Brown, S., Houghton, A.M., Trexier, M.C., Wisniewski, 1994; Fang et al., 2001; Intergovernmental Panel on Climate Change (IPCC), 2000; Wang et al., 2013). Forests store carbon in trees, litter and soils (van Kooten et al., 1999). Land use management practices such as conservation tillage and reforestation are becoming essential to sequester and store carbon to mitigate climate change impacts (Stringer et al., 2012).

The carbon sequestration rates for most of the land uses are calculated on the basis of long-term carbon stocks in those ecosystems. Tree species carbon sequestration rates are traditionally determined through use of allometric equations that approximate total carbon from estimates of living biomass. The living biomass is predicted from measurements of stem size for a given tree age, using a temperate species biomass equation such as $M = aD^b$, where a and b represent site specific growth parameters. D is diameter at breast height, and M is oven-dry weight (Gale et al., 2009; Ter-Mikaelian and Korzukhin, 1997). Carbon fixation rates by crops are those estimated for southern Quebec, i.e. under comparable climate conditions (Winans et al., 2015), based on C_P (product biomass in harvested plants equal to Yield*carbon content), C_s (carbon fixed in stubble residue including straw and litter fall), C_R (carbon in root biomass), C_E (carbon in root turnover and root exudates), C_i (carbon input to soil) and C_{is} (carbon storage in stable soil organic compounds) (Winans et al., 2015). We used C_{is} for estimating long term carbon storage in soils.

The net carbon sequestration values of natural and agricultural ecosystems are calculated using the avoided cost method:

$$U_{co_{2_i}} = S_i * P_c \tag{2.7}$$

where $U_{co_{2_i}}$ is the unit value of carbon sequestration for the ith land use (\$/ha/year); S_i is the carbon sequestration rate for the ith land use (tons CO2/ha/year) and P_C is the carbon price (\$/ton of CO2).

Carbon pricing (P_C) provides a way to account for the cost of greenhouse gas pollution and climate change (Dion and Laurent, 2012). Different Canadian provinces have proposed pricing schemes such as Alberta in 2007, where one option for emitters is to pay $15 per ton of carbon dioxide. In 2008, British Columbia introduced a carbon tax of $10 per ton of carbon dioxide that should rise to a maximum of $30 per ton in 2012 with an increment of $5 per year (Ministry of the Environment and Climate Change, 2015). In 2015, the province of Ontario entered the cap and trade program already in use by Québec (Canada) and California (US). For Québec and California, the minimum price was set to $10 (US$) per ton CO2 in 2013, increasing at 5% per year plus inflation which led to $15.84 (CAD) at the Quebec-California auction in August 2015 (Ministry of the Environment and Climate Change, 2015; Purdon et al., 2014). In our analysis, we used the 2015 price of $15.84.

2.3.4.3 Nutrient cycling

Natural ecosystems regulate their nutrient balances and tend to limit nutrient losses. Nutrient cycling maintains productivity of the land by keeping productive soils as they store and recycle their nutrients (Wilson, 2008a). The turnover of soil organic matter is the major pathway through which nutrients are made available again (Kennedy et al., 2011), while plant biomass production is the driver of nutrient uptake (Poorter et al., 2011). Agricultural land use affects the nutrient content (N, P, K etc.) of soils and changes their fertility and productivity (Uzoho et al., 2007). Agroecosystems are supplied with nutrients (fertilizers) which are ultimately lost to the nearby stream or to the atmosphere (Cadish and Giller, 1996; Reid et al., 2005). In agroecosystems, a substantial amount of nutrients is also removed by harvesting crops. The nutrient losses are countered by the application of synthetic fertilizers (Kennedy et al., 2011). The conceptual framework to contrast nutrient cycling in natural and agricultural soils is shown in Figure 2.2. The recommended fertilizer application rates in Ontario (OMAFRA, 2016) are used to estimate nutrient element inputs to agricultural lands.

Here, we focus on nitrogen, phosphorus and potassium (N, P, K) as the essential macronutrients. The unit values of nutrient cycling in different land use categories is calculated using the replacement cost method:

$$U_{N_i} = \left(N_{u_i} - N_{a_i}\right) * P_N \tag{2.8}$$

where U_{N_i} is the unit value of nutrient cycling for the *i*th land use ($/ha/year), N_{u_i} is the nutrient uptake rate of *i*th land use (kg/ha/year), N_{a_i} is the fertilizer application rate for the *i*th land use (kg/ha/year), $\left(N_{u_i} - N_{a_i}\right)$ is the net N, P or K uptake rate for the *i*th land use category, and P_N is the unit price ($/kg). The price of fertilizers, based on a farm input survey in Ontario (McEwan, 2015), is used as unit value of nutrient application (P_N).

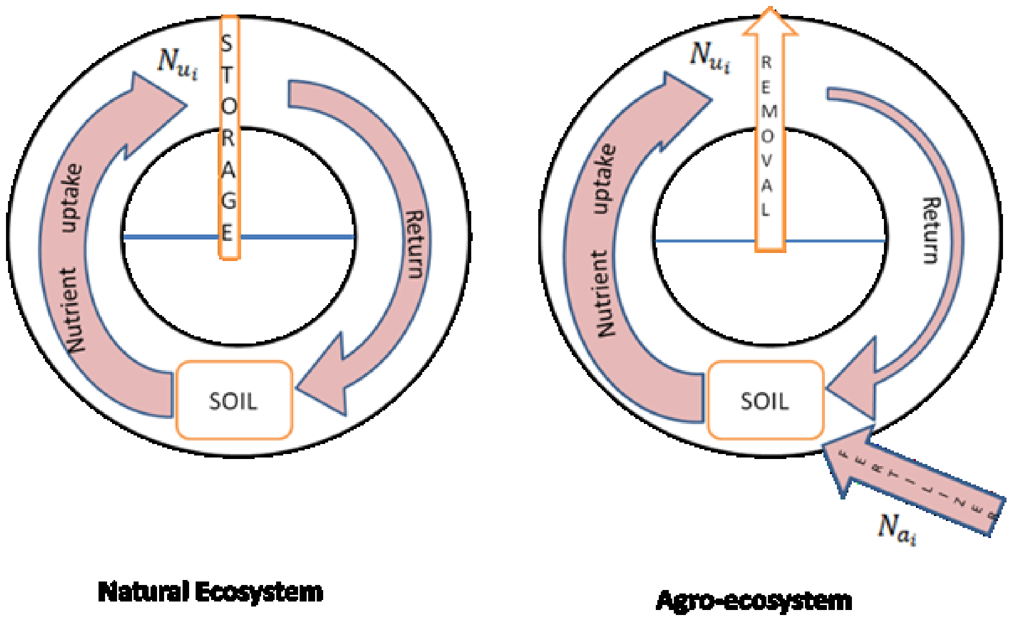

Figure 2.2: Conceptual framework for nutrient cycling in natural and agro-ecosystems.

2.3.5 Land use scenarios

We compare the values of the four ecosystem services in the Grand River watershed for the current (2015) land use to those of three hypothetical land use scenarios and their associated population changes. The scenarios are based on reconstructions for the years 1800 and 1900, and projections for year 2050. These scenarios help illustrate how the supply of ecosystem services and their economic values vary along a representative land use trajectory. Per capita water consumption and market rates for the valuation of selected ecosystem services are kept constant at their 2015 values in order to focus the comparative analysis on the role of land use and demographics. A GIS map of pre-European settlement (Butt et al., 2005) is used to construct the land use scenario for the year 1800 (see section 2.3.6.2). The year 1800 population is taken from the Historical Atlas of Canada: about 2600 people were living in the watershed, mostly concentrated in the lower watershed close to Lake Erie (Matthews et al., 1987). By year 1900 the watershed had only 5% forest cover left and a population of 179,000 (GRCA, 2014). The other land use types are adjusted according to this information using the current land use categories as baseline (see 2.3.6.1). The year 2050 scenario accounts for the projected population growth to 1.5 million for the watershed and assumes land use policies that prioritize reforestation and densification of urban areas (Section 2.3.6.4).

2.3.5.1 Consumptive services

In the scenarios, variations in water supply are only driven by changes in population, in analogy to studies in which past and future water demands are estimated (e.g., Gleick et al., 2005; Houston et al., 2003; Roo et al., 2012; Vandecasteele et al., 2014). Applying the current per capita water demand is consistent with the use of the current market price according to the concept of price elasticity of demand (Renzetti et al., 2015). Current water consumption per hectare in agriculture is applied to the agricultural land cover across the scenarios (Table 2.3).

2.3.5.2 Non-consumptive services

Each land use, except agriculture, is evaluated in the same way in all scenarios. Agricultural crops in the past yielded less biomass per hectare due to less intensive cropping practices and the absence of synthetic fertilizers. Therefore, in the 1800 and 1900 scenarios, crop production is reduced by the total factor productivity (TFP), which is the ratio of aggregate output quantities to aggregate input quantities and is used as a measure of agricultural productivity (Fantino and

Veeman, 1997). The total factor productivity (TFP) growth for the period from 1871 to 1921 (McInnis, 1986) is applied to the year 1900 scenario. For Canadian agriculture, TFP in the 1871-1921 period was on average 0.87% for high labor share and 0.77% for low labor share (McInnis, 1986). The TFP trend from 1990 to 2009 is extrapolated to obtain the biomass yield in year 2050. The TFP factor for year 1800 is similarly inferred by linear extrapolation back in time of the TFP trend reported for the period 1871-1921. We used an average TFP (for low and high labor share) of 0.82% per year for 1871-1921 (McInnis, 1986) and extrapolated it linearly from 1940 to 1900 because a significant change in TFP growth occurred from 1940 onwards (Dennis and Işcan, 2009). Uncertainties associated with these extrapolations are assessed below.

The total factor productivity (TFP) growth for the period 1871- 1921 (McInnis, 1986) is applied to the year 1900 scenario. For Canadian agriculture TFP from 1871 to1921 was 0.87% for high labor share and 0.77% for low labor share (McInnis, 1986). The TFP trend during the 1990-2009 period is extrapolated to obtain the biomass yield in year 2050. The TFP factor for year 1800 is similarly inferred by linear extrapolation of the TFP change between 1871-1921. From 1940 to 2009, TFP for crops in Ontario grew 1.21% per year. This yearly increase combines the effects of technical change (seed variety, genetic technology) (0.54), scale effects (intensive cropping practice) (0.63), technical efficiency change (0.08) and residual changes (-0.04) (such as measurement errors and changes in allocative efficiency) (Darku et al., 2016). We used an average TFP (for low and high labor share) of 0.82% per year for 1871-1921 (McInnis, 1986) and extrapolated it linearly from 1940 to 1900 because a significant change in TFP growth occurred from 1940 onwards (Dennis and Işcan, 2009). The crop productivity growth in Ontario for the last two decades (1990-2009) has slowed down, compared to the period 1940-1990, resulting in a TFP of 1.14% for this period (Darku et al., 2016).

Based on the above, the biomass yield of agricultural land use in the year 1900 is decreased by a factor of 3.40 $[(1.0121)^{69}*(1.0114)^{6}*(1.0082)^{40}]$, and for year 1800 by a factor of 7.70 $[(1.0121)^{69}*(1.0114)^{6}*(1.0082)1^{40}]$ based on the TFP of 0.82% per year between 1871and 1921 (McInnis, 1986). Using an incremental factor based on TFP for the target land use (2050) assumes that agricultural yield will increase through technical efficiency and not through increase in fertilizer application. The agricultural biomass production in year 2050 is increased by a factor of 1.49 $[(1+0.0114)^{35}=1.49)]$ by applying the TFP of 1.14% (as observed for 1990-2009) to the next 35 years (from 2015 to 2050).

2.3.6 Land use

The current land use (2015) is used as the baseline for comparison, and the same land use categories are adopted in the past and target land use scenarios. These scenarios are designed to account for the impact of land use and population changes while keeping other drivers (e.g., water consumption per capita, market price) fixed to those of the baseline scenario level (similar to Fezzi et al., 2011). Intensively human-modified land uses (i.e. urban) lack the capacity to generate a net supply of ecosystem services (Burkhard et al., 2009) and land use proxies would be difficult to apply in urban areas due to the emergence of new land use categories such as new housing types, new open or built up space and new surface materials (Kremer et al., 2016). Similarly, the built-up (residential, industrial, commercial) and extraction (roads, beaches, bedrocks) land uses do not contribute significantly to our selected ecosystem services and they are, therefore, not analyzed for their values. Considering current land use as baseline, the areas devoted to golf courses are estimated from population size for the other three land use scenarios and included in the area of pasture/sparse forest due to comparable functioning for the selected ecosystem services (Caldwell, 2013).

2.3.6.1 Current land use

The 2015 distribution of land use areas for the whole watershed (Table 2.2) is based on GIS data layers provided by the Grand River Conservation Authority (GRCA). This dataset provides the most detailed information on different agricultural land use categories in the watershed (Environment Canada, 2013). It is updated for wetlands based on the latest information for this particular land use categories (GRCA, 2015). This update changes the 1999 share of ~0.5% wetlands to 9.5% in 2015, based mainly on reclassifying open waters as wetlands (Figure 2.1). The latter wetland coverage is corroborated by the Southern Ontario Land Resource Information System (MNR, 2008), which shows ~10% wetland coverage. The other major land use categories are: residential built-up (3%), industrial built-up (1.5%), agriculture (65.5%) forest (10%), pasture/sparse forest (8%).

2.3.6.2 Pre-European scenario (1800)

A GIS map for reconstructed pre-European land use in southern Ontario (Butt et al., 2005) is used to delineate the pre-settlement land use. This map is derived from original land survey notes during European settlement from 1798 to 1850. The map contains land use data for 79.6% of the watershed area; the other 20.4% of the total area is un-surveyed or has missing data. We assume that the proportions of land use categories in the 79.6% of documented area of the watershed can be extrapolated to the missing

20.4%. The reconstructed land use scenario for pre-European settlement is composed of deciduous forest (77%), coniferous forest (1%), mixed forest (6%), wetlands (15.15%) and cleared lands (0.85%). The cleared lands are then further distributed among residential and agricultural land uses (Table 2.2).

2.3.6.3 Year 1900 scenario

The hypothetical 1900 land use is based on limited data and information (GRCA, 2004). Lands in the Grand River Watershed were rapidly cleared for farming and urbanization, hence reducing tree cover to 5% by the year 1900 (Shifflett, 2014). Therefore, forest cover is decreased to 5% keeping the relative change in sub-categories proportional to the forest sub-categories in the current land use. The agricultural land use is increased to cover the area cleared from forests and wetlands while keeping relative changes in sub-categories proportional to the current ones (Table 2.2).

The changes in population density are based on population data from1971 to 2011 (Statistics Canada, 2016), and extrapolated back to infer a density of 2550 persons/km^2 or 25.5 persons/ha in 1900. The urban population of Canada was 37 % in 1900 (Statistics Canada, 2016). We apply this percentage to the total population of the watershed in the year 1900 to roughly estimate the size of the urban population. The estimated urban area in 1900 is then 2596.5 ha. The industrial land use is derived from the baseline scenario (i.e., the current land use) and decreased proportionately by the change in population between 1900 and 2015.

Wetland coverage is kept at the 10% baseline given a report on the Grand River drainage authored in 1932 (Finlayson Report of 1932) which mentions the existence of a sizeable amount of swamps and marshes in the region (Shifflett, 2014). The 1900 open water area is reduced to 50% relative to the baseline to account for the seven dams and reservoirs that were built in the watershed between 1942 and 1946 (GRCA, 2015).

2.3.6.4 Year 2050 scenario

According to a recommendation of Environment Canada, a minimum coverage of 30% forest and 10% wetlands enables the healthy ecological functioning of a watershed (Environment Canada, 2013). These estimates are based on minimum requirements for species richness and aquatic ecosystem health (Environment Canada, 2013; Shifflett, 2014). We, therefore, impose a forest cover of 30% in the future scenario. Pasture/sparse forest areas and bare agricultural lands are assumed to be converted to forest to meet the 30% target. For the 2050 urban areas, we assume a higher population density to account for future, less energy and carbon intensive, cities (Angel, 2012; United Nations Human Settlements

Programme, 2013). Using an optimal urban density of 60 people/ha (United Nations Human Settlements Programme, 2013), and a projected population of 1.5 million (Shifflett, 2014), we calculate a future urban residential area of 25000 ha in the watershed. We add an industrial area of 14800 ha, that is a 50% increase above the baseline scenario, to match the population increase. Open water and wetlands areas are kept the same as there are no additional dams planned to be built in the watershed in the near future. All other land uses are distributed proportionally to the 2015 baseline distribution (Table 2.2).

Table 2.2: Land use scenarios for the Grand River watershed. (Note: Built up and extraction land uses are not valuated for ecosystem services.)

	Area (x10^3 hectares)			
Land Use	**Pre-European (year 1800)**	**Year 1900**	**Current (year 2015)**	**Target (year 2050)**
Row crops	2.4	188.9	133.1	128.3
Small grains	1.4	113.1	79.7	76.8
Forage	2.2	180.8	127.4	123.2
Pasture/sparse forest	-	53.9	55.7	21.1
Dense forest (Deciduous)	523.0	18.1	35.7	49.2
Dense forest (Conifer)	6.4	6.0	11.7	16.1
Dense forest (Mixed)	41.2	10.0	19.5	26.8
Plantation (Mature)	-	1.7	5.3	112.0
Wetlands	103.2	64.2	64.2	64.2
Open Water	-	3.4	8.5	8.5
Bare agriculture fields	-	35.8	106.0	10.1
Built-up (residential/industrial)	0.1	3.4	29.8	39.8
Extraction (roads/beach/bedrock)	-	0.6	3.3	3.76
Total	679.8	679.8	679.8	679.8

2.3.7 Impact of ecological scarcity on the value of non-consumptive ecosystem services

Ecosystem services can become highly non-linear in the vicinity of certain critical (tipping) points. For example, the relationship between flood severity downstream and tree density upstream can change dramatically when the latter crosses a critical threshold as a result of deforestation. Therefore, linear valuation methods may not be acceptable close to these critical points or thresholds (Farber et al., 2002). However, most valuation studies of land use scenarios in the literature (e.g., Dupras et al., 2016; Tolessa et al., 2016) do not reflect ecological scarcity in the economic analysis of ecosystem services.

We derive weightage factors for the past (1800 and 1900) and target (2050) land use scenarios considering the current (2015) land use scenario as the baseline. The weightage factors are the ratios of the unit values of natural land use (wetlands and forests) in the past or future scenarios to the unit values in the baseline scenario. As the percentage of natural land use increases (and the ecological scarcity decreases) the unit value can decrease, and vice versa. Following Ghermandi et al. (2010), the weightage factors are obtained from the standardized wetland value for ecosystem services plotted against wetland abundance within a 50 km radius. These factors are then applied to both wetlands and forests in the three hypothetical land-use scenarios. The use of the same factors for both land use categories is justified by their very similar values for the three non-consumptive ecosystem services.

2.4 Results

2.4.1 Water supply

The Region of Waterloo supplies bulk water to the municipalities of Cambridge, Waterloo, Kitchener, Wilmot and Woolwich at a wholesale rate (*U_w*) of \$ 1.006 cents per cubic meter (Region of Waterloo, 2016). The wholesale rate does not include the cost of the distribution network and its maintenance and therefore, represents a similar level of services as supplied by natural ecosystems.

For 2015, the total water consumption (Q_w) in the Grand River watershed is 152 Mm^3 per year distributed over municipal (61%), agricultural (11%), industrial (14%), rural domestic (4%), and other uses (e.g., dewatering, aquaculture, 10%) (GRCA, 2015). Per capita and agricultural plus industrial unit water consumption rates are calculated for the baseline scenario based on population and land use area, respectively. Multiplying per capita and unit water consumption with past and projected population numbers (see section 2.3.3), and land use areas, the water uses for the hypothetical past and future scenarios are computed (Table 2.3).

2.4.2 Water filtration

2.4.2.1 Sediment and phosphorus delivery rates (SDi, PDi)

Relative values of water filtration are obtained by subtracting the sediment delivery rate (*SDi*) of each land use category by the maximum watershed or regional sediment delivery rate (SD_{max}). Here, based on regional data, horticulture yields the maximum sediment delivery rate of 1.71±0.52 tons/ha/year (van Vliet et al., 1978)). For phosphorus delivery rates (*PDi*), we assign the maximum rate of 0.89±0.45 kg/ha/year reported for urban areas (Donahue, 2013; Hore et al., 1973; Hutchinson Environmental Sciences Ltd., 2012; Shaver et al., 1994; Winter, 1998).

2.4.2.2 Sediment and phosphorus removal costs (SR_c , PR_c)

The unit cost of sediment removal (SR_c) from SWMFs varies widely due to variation in site and catchment area characteristics and the available disposal options. The costs reported by the Toronto and Region Conservation Authority (2016) for ten SWMFs range from \$58/m^3 to \$265/m^3, with an average of \$170±78/m^3. Based on three years' performance of 12 WPCPs, one WWTC, and one STP, the average operating cost for total phosphorus (TP) removal is \$19±13 per kg expressed in 2015 CAD (XCG Consultants and Ltd., 2010). The unit costs for sediment and phosphorus removal are applied to unit delivery rates (Table 2.3) to compute unit values for water filtration for major land use categories in the watershed (Table 2.4).

Table 2.3: Total water use (million m^3) for domestic, agricultural and industrial activities, in the four land use scenarios. The computations are based on current annual consumption rates of 94.13 m^3/person (domestic), 78.70 m^3/ha (agriculture) and 2130.56 m^3/ha (industry), and the current wholesale rate of \$1.006/m^3.

	Pre-European (1800)	**Year (1900)**	**Current (2015)**	**Target (2050)**
Population	2,600	179,000	985,000	1,500,000
Water uses (Mm^3)				
Municipal	0.25	17.49	92.72	146.40
Agricultural	0.31	25.05	38.00	43.20
Industrial	-	4.02	21.28	31.53
Total water use (Mm^3)	0.56	46.56	152.00	221.13
Total value (million $/year)	0.56	46.84	152.91	222.45

Table 2.4: Sediment and phosphorus delivery rates and unit values for major land use categories in the watershed using equations (2.3) and (2.4). (See section 2.3.4.1).

Land Use	Sediment delivery rate, SD_i (t ha^{-1} $year^{-1}$)	Unit Value, U_{SFi} (\$$ha^{-1}$ $year^{-1}$)	Phosphorus delivery rate, PD_i (Kg ha^{-1} $year^{-1}$)	Unit Value, U_{PFi} (\$$ha^{-1}$ $year^{-1}$)
Row crop	0.70±0.55[a]	92±96	0.57±0.42[c,d]	11±11
Small grains	0.72±0.50[a]	90±95	0.62±0.68[d,e]	12±15
Forage	0.55±0.45[b]	108±88	0.14±0.06[f]	3±2
Pasture	0.08±0.06[a]	148±78	0.21±0.28[c,g]	4±6
Forest	0.04±0.03[a]	152±78	0.11±0.07[c,g]	2±2
Wetland	0.04±0.03[a]	152±78	0.05±0.02[h]	1±1
Open water	-	-	0.30±0.02[c,h]	6±4

[a](van Vliet et al., 1978); [b](Fox, G. and Dickson, 1990); [c](Winter, 1998); [d](Donahue, 2013); [e](Jeje, 2006); [f](Hore et al., 1973); [g](Shaver et al., 1994); [h](Hutchinson Environmental Sciences Ltd., 2012)

2.4.3 Carbon sequestration

We estimate an average biomass yield of 7.9 Mg/ha/year for grain corn (row crop) (mean of low, 6.5 Mg/ha/year, and high yields, 9.3 Mg/ha/year, Winans et al., 2015). As the variation of maximum and minimum yield to average yield is about 20%, we apply the same range to the CO_2 sequestration rate. This corresponds to an average CO_2 sequestration rate of 3.28±0.7 tons of CO_2/ha/year. A similar approach is used for other agricultural land uses (forage and small grains) to calculate CO_2 sequestration rates (Table 2.5). For the past scenarios (years 1800 and 1900), crop biomass production was low, which is accounted for by applying TFP to these scenarios. Consequently, the carbon sequestration rates for crops are decreased in these scenarios (see section 2.3.5.2).

Different types of wetlands (bogs, fens, marshes, swamps) exhibit variable net primary production rates (NPP) (Campbell et al., 2000; Vitt et al., 2001). Here, we impose the average carbon sequestration rate proposed by Mitsch and Gosselink (2015) for temperate freshwater wetlands. Open water ecosystems can provide CO_2 storage through the burial of organic matter (e.g., Delille et al., 2014), but inland waters can also be a source of CO_2 (Raymond et al., 2013), as in the case of hydroelectric reservoirs (Huttunen et al., 2002). Pelletier (2014) also concluded that the open water pools in peatlands are a source of CO_2. Detailed information on open waters CO_2 fluxes is not available for the Grand River watershed. Until this knowledge gap is filled, we assume a net CO_2 sequestration of zero for open water bodies, i.e. production and respiration cancel each other.

In their study, Thevathasan and Gordon (2004) propose that the typical silvipasture (pasture combined with trees) has a density of 111 trees per hectare. They further quantified above- and below-ground carbon sequestration based on the destructive sampling of trees (thirteen-years old poplar and Norway spruce). From their results, the average CO_2 sequestration rate for silvopastoral systems with fast-growing tree species plus monoculture pasture system is estimated at 6.6±4.6 t CO_2/ha/year. This is the value we use for the pasture/sparse forest land use category.

2.4.4 Nutrient cycling

2.4.4.1 Net nutrient uptake rates (Nu-Na)

The nutrient uptake rates (nitrogen, phosphorus and potassium) for forests are based on the study by Cole and Rapp (1992) with most of the sites for temperate coniferous and deciduous forests are located in the USA. The age of forest stand at these sites varies from 50 to 450 years. The sites from this study are comparable to the temperate forests in the Grand River watershed. The nutrient requirement for mixed forest is taken as the average of deciduous and coniferous forest. Further, we calculated nutrient uptake rates for sparse forest/pasture based on Bermuda grass and 9% of mixed forest depending on the tree density (Darst et al., 1996; OMAFRA, 2016).

According to a study on wetland plants, the concentration of nitrogen (N) in 30 different species is 1.5±1 % and the concentration of phosphorus (P) is 0.3±0.2 % dry weight biomass for 40 different species (McJannet et al., 1995; Campbell et al., 2001). We apply these percentages to calculate N and P uptake by wetlands. Wang and Moore (2014) report average C:N:P:K mass ratios of 445:14:1:9 for five functional plant types at the Mer Bleue bog in eastern Ontario. We use these stoichiometric ratios to estimate the potassium (K) uptake in wetlands.

The mean above ground NPP for fens and bogs is 337±142 g m^{-2} yr^{-1} and for marshes and swamps 924±463 g m^{-2} yr^{-1} (Campbell et al., 2000). Amongst the total wetland area in the Grand River watershed, 99% are marshes and swamps, while fens and bogs make up the remaining 1% (MNR, 2008). These percentages are used to assign the unit value for wetlands in the watershed.

Net nutrient (N, P, and K) uptake rates (N_u - N_a) are computed as the gross nutrient uptake rates (N_u) minus the nutrient application (fertilizers) rates (N_a), as described by equation 2.7. Corn, wheat and alfalfa are representative for row crops, small grains and forage, respectively, in the Grand River watershed (Scott, 2006). In our estimations, we assume that nutrient deficiencies for crops are met by applying the amounts of synthetic fertilizers recommended by the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA).

Table 2.5: Net CO_2 sequestration rates for current (2015) land use in the watershed. To calculate the corresponding economic values, we apply \$15.84 per ton of CO_2 sequestered (See section 2.3.4.2)

Land use	**Net sequestration rates, S_i (t CO_2 ha^{-1} yr^{-1})**	**CO_2 sequestration value (\$/ha/year)**
Row crops	3.3±0.7[b]	52±13
Forage	1.75±0.4[b]	28±8
Small grains	1.54±0.1[b,c]	24±2
Pasture/sparse forest	6.6±4.6[f]	105±73
Forest (Deciduous)	84±10[d,e]	1330±160
Forest (Coniferous)	34±4[d,e]	540±60
Forest (Mixed)	54±7[d,e]	855±110
Plantation (Mature)	58±4[g]	910±70
Wetlands	10.2±1.5[a]	162±24

[a](Mitsch and Gosselink, 2015); [b](Winans et al., 2015); [c](Stellacci and Caliandro, 2007); [d](Gale et al., 2009); [e](Thomas and Martin, 2012); [f](Thevathasan and Gordon, 2004); [g]Average of the three forest types.

The net nutrient uptake rates for other land uses were calculated in a similar way (Table 2.6). Note that negative uptake values imply that the application rate exceeds the uptake rate. Further note that fertilizer application rates for years 1800 and 1900 are assumed to be zero. The current optimal fertilizer application rates are applied to the target year 2050 land use scenario because increase in TFP will be achieved through technical efficiencies and not increased fertilizer application (see section 2.3.5.2).

2.4.4.2 Nutrient pricing (P_N)

Nutrient pricing for N, P and K is based on the market rates of commercial fertilizers in Ontario. An average price of \$1.37/kg N is based on the cost of typical nitrogen fertilizers: anhydrous ammonia (82-0-0), urea 46%, nitrogen solution (UAN) 28%, and ammonium nitrate 34%. The average price of \$2.94/kg P is based on the cost of typical phosphorus fertilizers: mono-ammonium phosphate 11-52-0, di-ammonium phosphate 18-46-0, triple superphosphate 0-46-0. An average price of \$1.12/kg K is based on the cost of typical potassium fertilizer: muriate of potash (60%) (McEwan, 2015). Applying equation (2.8), the unit values for nutrient cycling for major land use categories are the obtained (Table 2.6).

2.4.5 Whole watershed valuation

The unit values of the four ecosystem services are given in Table 2.7 for current conditions and in Table 2.8 for the hypothetical land use scenarios. The total watershed value of the four combined ecosystem services decreases from a maximum of \$970±95 million per year for the pre-European land use scenario to a minimum of \$240±30 million per year for the year 1900 land use scenario. The value increases back to \$625±37 million per year in the 2050 scenario.

Table 2.6: Net nutrients (N, P, K) uptake rates and values for the current (2015) land use in the watershed.

Land use	N uptake, (N_u-N_a) (kg/ha-year)	P uptake (N_u-N_a) (kg/ha-year)	K uptake (N_u-N_a) (kg/ha-year)	Nutrient cycling value (U_N) ($/ha/year)
Row Crop	35±60[a]	-8 ±15[a]	42±56[a]	70±115
Small grains	-15±13[a]	-4±0.5[a]	45±1[a]	18±18
Forage	-12±50[a]	-4±9[a]	-14±30[a]	-42±80
Pasture/sparse forest	170±1[g,h]	28±0.1[g,h]	190±0.5[g,h]	530±1.5
Forest (Deciduous)	60±12[b]	4 ±1[b]	50±5[b]	145±20
Forest (Coniferous)	40±12[b]	6±3[b]	32±9[b]	110±22
Forest (Mixed)	50±10[b]	5±1.5[b]	40±5[b]	130±15
Wetlands	101±78[c,d]	10±6[c,d]	58±55[c,d]	235±125
Open Water	475±70[e,f]	50±5[e,f]	100±15[e,f]	900±100

[a](ERIN Consulting Ltd., 2006); [b](Cole and Rapp, 1981); [c](Wang and Moore, 2014); [d](Campbell et al., 2000); [e](Fong et al., 2004); [f](Ho et al., 2003); [g](Darst et al., 1996); [h](OMAFRA, 2016)

Table 2.7: Unit values of the three non-consumptive ecosystem services (U_w, U_{co2}, U_N) and total unit values (Vi) of major land use categories in the watershed for current (2015) land use using equations (2.3 to 2.8).

Land use	**Unit values for ecosystem services ($/ha/year)** Water filtration (U_w)	CO$_2$ sequestration (U_{co2})	Nutrient cycling (U_N)	**Total Unit Value (V_i) ($/ha/year)**
Row crops	98±97	52±13	70±115	220 ± 150
Small grains	95±96	28±8	18±18	140 ±100
Forage	120±90	24±2	-42±80	100 ± 120
Pasture/sparse forest	160±80	105±73	530±1.5	795 ± 110
Dense forest (Deciduous)	166±80	1330±160	145±20	1640±180
Dense forest (Conifer)	166±80	540±60	110±22	815±102
Dense forest (Mixed)	166±80	855±110	130±15	1150±135
Plantation (Mature)	166±80	910±70	130±10	1205±105
Wetlands	168±80	162±24	235±125	565 ± 150
Open Water	11±11	-	900±100	910±100
Bare agriculture lands*	10±55	35±5	15±45	155 ± 70

*Bare agricultural lands are given the average value of row crop, small grains because these lands have been or will be under similar use as agricultural lands.

Table 2.8: Unit values of three non-consumptive ecosystem services (U_w, U_{co2}, U_N) for the past (years 1800 and 1900) and future (year 2050) agricultural land uses in the watershed adjusted for TFP (see section 2.5.2).

	Total Unit Value, Vi ($/ha/year)		
Land use	Year 1800	Year 1900	Year 2050
Row crops	115±98	135±100	280±200
Small grains	100±96	110±95	165±100
Forage	120±90	115±95	95±150
Bare agricultural lands	110±55	120±55	180±85

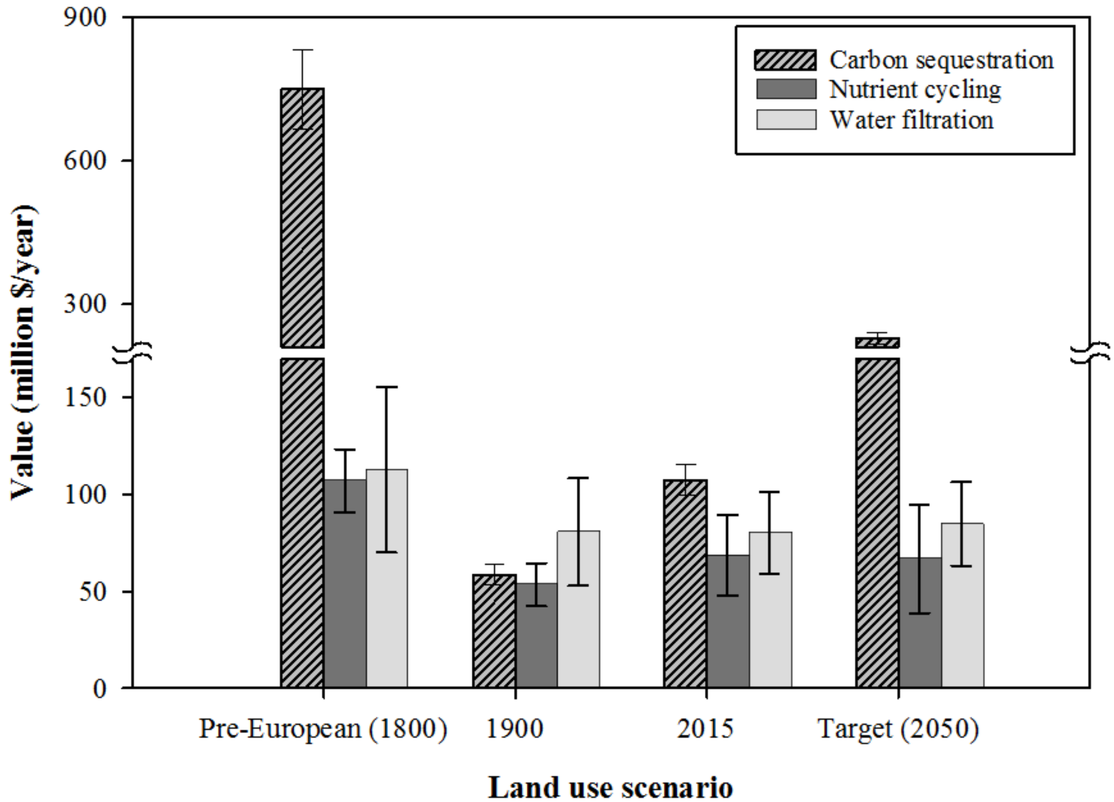

Figure 2.3: Total values of three non-consumptive ecosystem services for four land use scenarios (error bars show standard deviations).

2.4.6 Values of consumptive versus non-consumptive ecosystem services

The combined value of the non-consumptive services considered (carbon sequestration, nutrient cycling and water filtration) is highest for the pre-European land use scenario and lowest for the year 1900 (Figure 2.4). From 1900 onwards the value of the consumptive services increase again. By contrast, the value of the consumptive service (water supply) shows a continuous increase from pre-European to future target conditions (Figure 2.4) due to increasing population and industrial growth across the scenarios.

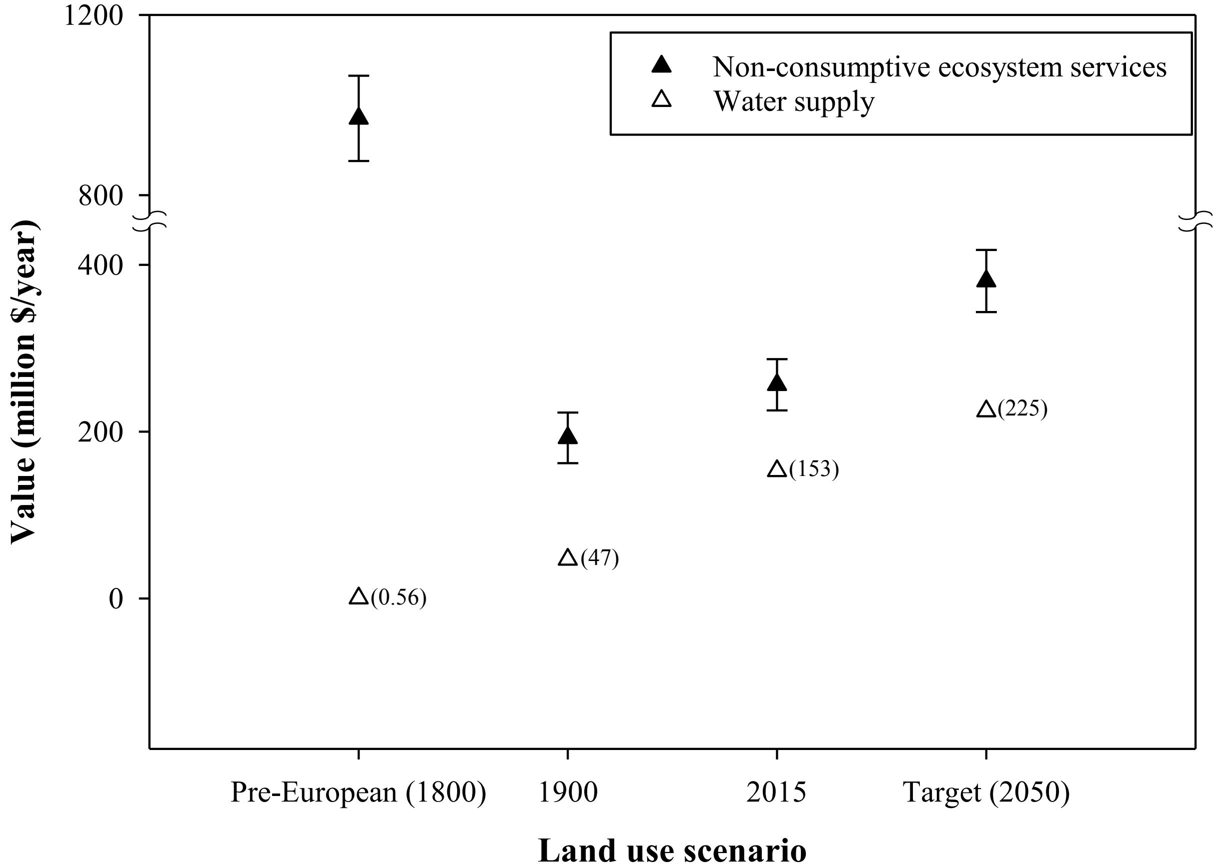

Figure 2.4: Consumptive (water supply, values given in brackets) and non-consumptive ecosystem services (carbon sequestration, nutrient cycling, water filtration) for four land use scenarios (error bars show standard deviations).

2.5 Discussion

2.5.1 Sensitivity analysis on land use and TFP

We performed a sensitivity analysis on land use area variations for the current land use, and on TFP variations for past and future land use scenarios (see supplementary material for details). The land use sensitivity analysis reveals a maximum increase of 8.5 % in the total value of three non-consumptive ecosystem services when the deciduous forest area is increased by 50%. Among all natural land uses, deciduous forest has the highest unit value, and has larger area than other forest types in the current (2015) land use. Thus, the effects of 10% and 50% increases in all other land uses are lower than those for deciduous forest. Using a TFP decrement factor for the 1800 scenario equal to that of year 1871 only increases the total value of non-consumptive ecosystem services by less than 1% compared to the decrement factor obtained by linear extrapolation of TFP between 1871 and 1800 (see the supplementary material for details).

Table 2.9: Sensitivity analysis for variations in land use areas in the Grand River watershed.

Land Cover	Area (ha)	Best estimate ($/ha/year)	Percent change in total value with increase in one and proportional decrease in all other land uses		Percent change in total value with 10% increase in one land use area only
			10% increase	50% increase	
Row crops	133,082	220	-1.25	-6.23	1.07
Small grains	79,662	140	-0.94	-4.71	0.41
Forage	127,389	100	-1.88	-9.40	0.47
Pasture/sparse forest	53,578	795	0.80	3.99	1.56
Dense forest (Deciduous)	35,722	1640	1.69	8.45	2.15
Dense forest (Conifer)	11,731	815	0.17	0.86	0.35
Dense forest (Mixed)	19,497	1150	0.54	2.68	0.82
Plantation (Mature)	5,305	1205	0.15	0.77	0.23
Wetlands	64,278	565	0.04	0.22	0.17
Open Water	8,475	910	1.28	6.39	2.15
Bare agriculture fields	106,029	155	-1.25	-6.23	0.60

2.5.2 Sensitivity analysis on ecological scarcity

In our analysis, values of non-consumptive ecosystem services show a strong positive correlation with the natural ecosystem (forest and wetlands) size in the watershed. Therefore, we perform a sensitivity analysis to evaluate the impact of natural land use scarcity on the value of the watershed. By applying the derived weightage factors (see section 2.3.7), the total values of three non-consumptive ecosystem services in the watershed are estimated at \$720±70, \$197±30 and \$360±36 million per year for the pre-European (1800), year 1900 and target (2050) land use scenarios, respectively. Compared to the original values, these estimates show a decrease by 25% in the value of pre-European scenario, and 3% in the value of target land use scenario. In both of these scenarios, natural land use (forest and wetland cover) is higher than in the current land use scenario. In contrast, the value of year 1900 land use scenario increases by 2% because of a lower forest cover (5%) compared to the current land use scenario.

2.5.3 Comparison to existing studies

In this study, we estimate the value of four ecosystem services including one consumptive (water supply) and three non-consumptive services (water filtration, CO_2 sequestration, nutrient supply). The water supply is valuated using the market price method whereas the non-consumptive ecosystem services are estimated based on land use area and unit values. The unit values are largely based on local and regional data, in line with Pandeya et al. (2016) suggestion that valuation cannot be policy relevant unless it integrates the local data, knowledge and socio-ecological approaches. The cost-based approaches are used to translate assessments of different social (e.g., water use) and ecological (e.g., carbon sequestration) benefits into the same unit of currency and illustrate the changes in value of ecosystem services with changing land use scenarios.

The analysis of the past, current and future scenarios illustrate the change in the capacity of ecosystems to deliver ecosystem services. In this study we show that land use dynamics have major impacts on the value of non-consumptive ecosystem services within the watershed. The value of the three non-consumptive ecosystem services drastically decreases with a reduction in the size of natural ecosystems, here primarily forest. The total value of four selected ecosystem services decreases from \$980±95 million per year in the pre-European (1800) scenario to \$245±30 million per year in the year1900 scenario due to the decline in the forest cover and the expansion of agriculture. Unlike non-consumptive ecosystem services, the value of water supply (a consumptive ecosystem service) increases along these scenarios as water consumption in the watershed increases in time. These land use scenarios are meaningful in gauging the impact of different drivers (here, land use change, human activity and population) in the watershed on the value of ecosystem services. The restoration of natural land cover has a positive impact on the value of three non-consumptive services whereas increasing anthropogenic activity has a positive impact on the value of the consumptive ecosystem service.

Water supply is valuated using the market price method; market price in this analysis is the wholesale rate for bulk water supply to the municipalities located within the watershed. The unit value is applied to the total water consumption in each scenario to infer the value of water supply. This is a direct and robust approach to value water supply service across scenarios in The Grand River watershed. Nevertheless, other approaches have been used. For example, a study on the neighboring Credit River watershed (Table 2.1) valuated water supply based on groundwater use and applying the replacement cost of pumping water from Lake Ontario (Kennedy and Wilson, 2009). The valuation study of British Columbia's lower mainland used bottled water price as replacement cost to value water supply service (Wilson, 2010). The bottled water price is among the more expensive alternatives and can be expected to yield higher values for water supply. However, Kuuluvainen (2002) suggested that approximate non-market valuation methods should not be used for an ecosystem service that can be linked to a direct market value. Here, we use a direct valuation method for a similar level of service provided by natural ecosystems.

The value of water filtration service is based on the relative sediment and phosphorus reduction by the land use categories in the watershed. The sediment removal cost is adopted from the local storm water management facilities (SWMF). These facilities are selected to best reflect the functioning and level of service provided by natural ecosystems. Alternatively, a similar valuation study on Lake Simcoe watershed (Wilson, 2008a) applies a global value to valuate soil erosion control. Further, it values forest and wetlands for water filtration based on a correlation between reduction in forest cover and increase in water treatment cost in the watershed, whereas agriculture (40% of the watershed area) and pasture (7% of the watershed area) were left out of the study. The aforementioned valuation study of British Columbia's lower mainland uses a water filtration cost equal to 50% of the amount paid by households in the Greater Vancouver Water District (Wilson, 2010), thereby not accounting for the role of different land uses towards water filtration service. In our study, we valuate all major land use categories in the watershed based on their relative capacity to filter water.

For phosphorus reduction value, we rely on the relative phosphorus reduction capacity of land use categories in the watershed, coupled to estimates of the unit cost of phosphorus removal in a water treatment plants. Despite a number of studies on phosphorus management in the Grand River watershed, there is no reliable information on the unit cost of phosphorus removal from water. A recent Environment

Canada report (Hanna, 2015a) explores phosphorus management options (e.g., fertilizer application timing and methods, upgrading wastewater treatment plants) in the Grand River watershed. This report uses a phosphorus (P) management decision support system (PMDSS) which incorporates P removal in crop harvest and permanent sedimentation sites during high water events (e.g., spring freshet across a broad flat floodplain) (Hanna, 2015b). It estimates an average cost of $1750/ha for providing cover crops and $37/ha for providing buffer strips for phosphorus management which makes them expensive substitutes compared to water treatment plants. The average effectiveness of cover crops is 40% and of buffer strips is 70 % for reduction of particulate phosphorus with respect to the baseline case of surface application of solid manure in the fall. Further, the Environment Canada study did not account for natural land use categories in the watershed (e.g., forests, wetlands). In our analysis, we assessed all major land use categories based on their relative phosphorus reduction capacity. The capacities of forests and small grain crops to retain phosphorus are 76% and 30%, respectively, more than the base land uses. The PMDSS model yields the cost and benefits of P management alternatives (Hanna, 2015b) and upgrade costs of wastewater treatment plants, but did not represent the operational cost of trapping phosphorus. Here, we propose an average phosphorus removal cost ($/kg) based on 12 WPCPs, a WWTC and a STP, which operate in local watersheds. These facilities provide exclusive unit rates for removal of phosphorus from the water. The variation in the values of phosphorus filtration is due to variations in phosphorus delivery rates and economic cost to remove the phosphorus from the water. The latter depends on many factors such as treatment processes, technology, and phosphorus loadings to different treatment plants. Therefore, the values provided are indicative of the order of magnitude of the removal costs and more detailed estimates can be made for individual plants and technical upgrades. The relative unit values of water filtration, in our analysis, can be increased by reducing sediment and phosphorus delivery from different agricultural land uses through better management practices (e.g., changes in tillage practices, riparian buffer, and cover crops).

The total value of CO_2 sequestration service in the watershed is higher than the values of the other three ecosystem services for all scenarios except year 1900, and results in a maximum value of $750±80 million per year for the pre-European land use scenario. The unit rate for carbon sequestration is based on the proposed carbon pricing of $15.84 per ton in year 2015 (Government of Ontario, 2016) adjusted for 5% annual increase plus inflation. Therefore, the carbon sequestration value strongly depends on the local political decisions. In another Ontario study in the Credit River watershed, climate regulation (carbon sequestration and storage) service yielded the highest value among 10 valuated ecosystem services because the study use the UK estimate of the unit social cost of $63 per ton (CAD 2015) of carbon

(Kennedy and Wilson, 2009), which is four-fold higher than the carbon pricing used here. Similarly, the value transfer method yields a unit value of $68 (CAD2015/ha/year) for Barnegat Bay Watershed, while a global damage cost of carbon in the atmosphere was used to estimate a unit value of $43/ha/year (CAD 2015) for Lake Simcoe watershed. In all of these, and similar, studies, the unit value of carbon sequestration is strongly influenced by the carbon pricing rather than by the carbon sequestration rate of an ecosystem. Consequently, local context and local political decisions must be considered for the valuation of ecosystem services.

The unit value of nutrient cycling is based on nutrient uptake rates (N, P, K) by a land use category in the watershed and market prices of these nutrients in commercial fertilizers. Opposite to the existing valuation literature, we infer unit values for all major land use categories in the watershed for the nutrient cycling service. This approach shows that the unit values of nutrient cycling for agricultural land uses can also be increased by using optimal nutrient amounts for crops (fertilizer input). In contrast, the valuation study on Lake Simcoe watershed valuated only three land use categories (pasture, hedgerows/cultural woodland, orchards) for nutrient cycling and attributed a single value to them. In our analysis, nutrient cycling is the only services wholse value is less for target land use scenario compared to current land use scenario. This decrease in value is due to increased TFP for future scenario, which will result in removal of more nutrients from the system.

Many valuation studies use indirect approaches to value nutrient cycling. Dodds et al. (2008) tied nutrient cycling to erosion rates and valued it for damage cost of subsequent runoff of nitrogen fertilizer from the fields. Based on damage cost, they assigned a value of $ 1404/ha/year (CAD 2015) to forest nutrient cycling service, compared to our value of $128/ha/year (CAD 2015). Another study (Curtis, 2004) assigned an average value of $7.42 (CAD 2015) to nutrient cycling in Australian forests using three different models. This value of nutrient cycling accounts for nutrient storage and carbon sequestration. Similarly, Byström (2000) valuated wetland soils in Sweden for their fertility for nitrogen and placed a value of $3.65/ha/year (CAD 2015). This value is based on the reduction in leakage of nitrogen downstream of wetlands (Byström, 2000) and is significantly lower than our estimated unit value of $235/ha/year for wetlands. These differences reflect the different valuation approaches, different numbers of benefits considered, data availability and the data itself. For example, we valuated the nutrient cycling in wetlands based on uptake rates by wetland vegetation whereas Byström (2000) linked it with the opportunity cost of abatement in nitrogen leakage.

Most of the watershed valuation studies (Table 2.1) used the benefit transfer method and unit values of ecosystem services vary over wide ranges. The variation in unit values may be assigned to multiple inherent sources of uncertainty such as considered benefits for each service, the valuation methods, and valuation parameters in the methods (Boithias et al., 2016). The unit value also depends on land use types in a watershed. As seen in Table 2.1, most valuation studies at a watershed scale (e.g., Skykomish, Snohomish, McKenzie, Credit River) consider only natural land uses (e.g., wetlands, grasslands) for valuation of water filtration service, while agriculture land uses are neglected or assigned a zero value (Batker et al., 2010; Kennedy and Wilson, 2009; Schmidt et al., 2011; Schmidt and Batker, 2012). Some valuation studies even leave out the forest contribution to water quality as in the case of Snoqualmie Watershed (Table 2.1). The majority of these studies (Table 2.1) use the lowest and the highest unit value of an ecosystem service from the literature to value a watershed which leads to high uncertainty in the total value of the watershed. For example, the Skykomish watershed study adopts a low bound value (of US$ 140/hectare/year) for soil erosion control in forests from a global estimate (Costanza et al., 1997) where the willingness to pay method is typically used for valuation and a high bound value (of US$ 316/hectare/year) from a national level study where the unit value is deduced from soil loss and annual cost of soil erosion (Dodds et al., 2008). Likewise, the Skykomish watershed study uses a lower unit value of nutrient cycling in forests taken from Dodds et al. (2008) and an upper unit value taken from Costanza et al. (1997). In fact, Dodds et al. (2008) used the avoided cost and Costanza et al. (1997) the benefit transfer method for the unit values. This trend in picking values of the same service from different, and not necessarily comparable, studies in terms of valuation methods, scale, and considered benefits leads to increased uncertainty and high variation in the total values of ecosystem services. In contrast, our unit values are based on biophysical assessments, and any variation in the values is due to random error in parameters (e.g., biomass), which can be reduced by more measurements.

Moreover, in some valuation studies, a variety of methods are used to value different land use categories for the same service. This is evident in the case of the Peace River watershed (Wilson, 2014) where the value of water filtration service by wetlands is estimated using replacement cost method, and the same service provided by forest is valuated using avoided cost method. This inconsistency potentially brings in a high disparity in the values for the same service. Another problem is that these studies use unit values from other primary studies which imply big assumptions in deriving unit values. For instance, in the valuation study of the Mackenzie Valley watershed (Anielski and Wilson, 2010) where sediment reduction value for agricultural land use was taken from an original study (Olewiler, 2004); whereas the original study valuated the agricultural land use for sediment reduction based on the assumption that vegetation will reduce sediment by 70% compared to a traditional land use.

Despite all its shortcomings, the benefit transfer method is quick and thus often the less expensive to use. The valuation studies discussed here (Table 2.1) all use the unit value transfer, however, the application of value function transfer or meta-analytic function transfer may add to improve on the valuation estimates. The strength of these methods is that they allow differences to be controlled between study and policy sites (e.g., population, area of ecosystem) (Brander, 2007).

To compare the dispersion of our estimated local unit values with other regional studies, we calculated an average value and associated standard deviation from low and high unit values used in the regional studies. The coefficient of variation in the unit values of these regional studies (CV_1) is compared with the coefficient of variation in the local unit values (CV_2) used in our analysis. The comparison shows that the coefficient of variation of the local unit values (CV_2) is significantly lower than CV_1 of the other regional studies (Table 2.10). Therefore, this study considerably narrows the vagueness (range) of the unit values. On the other hand, the studies which assign a single value to an ecosystem service (e.g., Credit River watershed, Table 2.1) undermine the fact that valuation is not a fixed science. The valuation studies contain multiple inherent sources of uncertainty which must be addressed or, at the very least, acknowledged (Hamel and Bryant, 2016).

2.5.4 Limits of the study

The factors which are not taken into account are the marginal change in economic values over time (Dupras et al., 2016), societal preferences across scenarios, and spatial heterogeneity of different ecosystems in the watershed. The assumption of static water demand for valuation of water supply service is not true yet inevitable due to (i) the complexity in forecasting/predicting water consumption per capita for long-term past and future scenarios, and (ii) the lack of data availability for such analysis in the watershed.

In our analysis, the value of water filtration service provided by different land uses is based on their relative capacities to retain sediment and phosphorus with respect to a land use with maximum delivery rate in the region. However, we acknowledge that absolute sediment retention values of natural ecosystems (e.g., wetland and forests) for water filtration could be higher.

An additional factor which can impact the total value of ecosystem services is that there are different types of wetlands (e.g., swamps, bogs, marshes, fens) in the Grand River watershed (GRCA, 2016) which are aggregated and assigned an average value for their ecosystem services. The functions of these wetland types, and subsequent ecosystem services, depend on their unique structure and functioning (Turner et al., 2000). Therefore, valuing of these wetland types – one of the most valuable ecosystems – will affect the total value of ecosystem services in the watershed.

Another source of error can be the selection of valuation methods. Some valuation methods give higher values than others. For example, Gunatilake and Vieth (2000) showed that replacement cost provide 29% higher estimates of the cost of soil erosion than the productivity change method (Gunatilake and Vieth, 2000). The reliability of different valuation methods at watershed scale clearly needs further investigation.

Table 2.10: Comparison of coefficient of variation, CV (=standard deviation ÷ mean) in the unit values of ecosystem services between regional studies (CV_1) and our study (CV_2).

Valuation area	Ecosystem service	Land use	CV_1 (%)	CV_2 (%)
Skykomish watershed	Nutrient Cycling	Forest	75	20
McKenzie watershed	Nutrient Cycling	Pasture	140	0
	Erosion control	Forest	54	51
Snohomish watershed	Water Quality	Wetland	140	44
Southern Ontario	Nutrient cycling[1]	Wetland	72	53
	Nutrient Cycling[2]	Wetland	138	53
	Nutrient cycling[3]	Pasture	126	0
Lake Winnipeg watershed[1]	Carbon sequestration	Forest	134	12

[1](Voora and Venema, 2008); [2](Troy and Bagstad, 2010); [3](Olewiler, 2004). Other studies are enlisted in Table 2.1.

2.6 Conclusions

We present the economic valuation of four ecosystem services in the Grand River watershed based on local unit values. Local unit values integrate the characteristics of local ecosystems and market values. We provide evidence (Figure 2.4) that supports the argument that consumptive and non-consumptive ecosystem services need different economic approaches for their valuation.

The unit values of non-consumptive ecosystem services are based on the production function of all major land use categories in the watershed. The unit values reflect the local context to facilitate incorporation/inclusion of values of ecosystem services into decision making. The valuation of hypothetical scenarios shows that the change in the value of non-consumptive ecosystem services is proportional to the variations in the size of natural ecosystems. The value of carbon sequestration is higher than other non-consumptive ecosystem services which is entirely a reflection of a political choice at the provincial level.. In the Grand River watershed, values of three non-consumptive services varies proportional to the naturalness (natural land use) and value of a consumptive service depends on the rate of resource removal.

In short,

- Valuation reflects the local context because unit values ($/ha/year) are based on local data.
- Our model for comprehensive valuation accounts for all major land use categories (natural and agricultural) in the Grand River watershed, and improves the uncertainty in the unit values of ecosystem services.
- We put forth the evidence that different drivers affect the value of consumptive and non-consumptive services; therefore they need different economic approaches for their valuation.
- Carbon sequestration service has the highest value among the selected ecosystem services, particularly because of recently implemented carbon emission tax in Ontario.
- The degradation of natural land use (e.g., forest, wetlands) resulted in a decline in the value of selected non-consumptive ecosystem services in the watershed. However, restoration of forest led to a boost in the value of these ecosystem services.

Chapter 3

Economic valuation of sediment and phosphorus retention by different wetland types in southern Ontario, Canada

3.1 Summary

Wetlands are known for their water filtration functions. Although different wetland types differ in their filtration capacity, they are usually aggregated together in economic valuation studies. Here, we explicitly separate the valuation of the sediment and phosphorus (P) filtration services of four wetland types – bogs, fens, marshes and swamps – across southern Ontario, Canada. The areal extents of the four wetland types are derived from the Canadian Wetland Inventory (CWI) progress map, while the sediment accretion rate is used as the key parameter regulating the sediment and P filtration functions. Using available literature data, we develop regression models to relate the sediment accretion rate to wetland size. The models, however, only show weak positive correlations, hence justifying the use of average sediment accretion rates to compare the sediment filtration efficiencies. Combining representative soil phosphorus concentrations with the sediment accretion rates yields phosphorus retention estimates for each of the four wetland types. The replacement cost method is then applied to valuate the sediment and P filtration services. The unit values for both sediment and P retention decrease in the order marshes > bogs > swamps > fens. The total value of sediment removal plus phosphorus elimination by all wetlands in southern Ontario amounts to \$3500±1640 million per year of which about 80% is accounted for by swamps. We further assess the costs of different options to offset the P loads generated by the hypothetical removal of all natural wetlands. The results demonstrate that replacing the water filtration function of existing wetlands by engineered solutions is not cost-effective.

3.2 Introduction

Wetlands, one of the most productive ecosystems, provide huge economic benefits through a variety of functions (Lambert, 2003). The hydrological (e.g., flood water retention), biogeochemical (e.g., nutrient retention) and ecological (e.g., nursery plants) functions of wetland ecosystems supply socio-economic benefits (ecosystem services) (Brouwer et al., 1997). There is a huge diversity (hydrologic, topographic, and geomorphic characteristics) in wetlands which is observed as wetland types (Warner and Rubec, 1997) and the functions of these wetland types depend on the uniqueness of their structure and characteristics (Turner et al., 2000).

In many areas of the world, ecosystem services have declined due to draining of wetlands (Zedler, 2003). Since year 1900, fifty percent of the wetlands are already lost worldwide (Ducks Unlimited Canada, 2010) and 85% of the total wetland losses in Canada are due to agriculture reclamation (Hotte et al., 2009). Similarly in southern Ontario (Canada), about 68% of wetlands are converted to other uses since 1980 (Ducks Unlimited Canada, 2010).

Wetlands have long been recognized for their key function of depurating pollutants from water due to their location on landscape (Gopal and Ghosh, 2008). Increased sedimentation and nutrients enrichment are big threats to the water quality in aquatic systems (Dordio et al., 2008). Therefore, the role of wetlands in improving water quality is the primary argument for their preservation (Dordio et al., 2008). Freshwater wetlands trap sediment and sequester nutrients (Craft and Casey, 2000). Wetlands filter water through physical (sedimentation), chemical (adsorption, precipitation, chelation) and biological (plant uptake, decomposition, mineralization) processes (Gopal and Ghosh, 2008). The water filtration service of wetlands is well known but their capacity to purify water is largely unassessed (Dordio et al., 2008).

Sediment and other nutrients (such as phosphorus) concentrated in sediment are water contaminants (Fennessy et al., 2008) and deteriorate water quality. Sediments enter wetlands through water inflows, rainfall and dryfall (wind-blown dust, ash, pollen). One of the major functions of wetlands in improving water quality is accumulation/retention of sediments (or sediment deposition) and nutrients from water and this function is the result of specific characteristics and complicated internal processes (Kadlec and Wallace, 2009) such as particle size and texture, residence time, wind and wave action, flocculation and vegetation, hydroperiod length, flooding and characteristics of stream flow (Fennessy et al., 2004; Kidd et al., 2015; Reddy et al., 1999; Settlemyre and Gardner, 1977). Sediment deposition depends on wetland type (Loaiza and Findlay, 2008), and some wetland types are more efficient at sediment and nutrient retention than others (Bruland, 2008). The effectiveness of wetlands in retaining sediment and nutrients also depends on watershed size, land use and their connectivity to rivers and streams (Craft and Casey, 2000). The sediment accretion in wetlands is heavily affected by the human activity within a watershed, such as in the case of Murray–Darling Basin in Australia where sedimentation rates doubled after the European settlement and became eighty times the mean rate of the Late Holocene (Gell et al., 2009).

Sediment accretion is a net balance between the sediment deposition and removal process (Neubauer et al., 2002) and is an important indicator of functioning of restored wetlands (Takekawa et al., 2010). It is a long term process and is a useful repository of nutrients and pollutants in wetland systems (Keller and Knight, 2004). The sediment accretion rate in wetlands is critical for providing ecosystem services, especially related to water quality (Bhomia et al., 2015). Sediment accretion rate is influenced by organic and inorganic material, vegetation, species composition, elevation and flooding patterns (Cahoon and Turner, 1989; Goodman et al., 2007; Gosselink and Turner, 1978; Martinez, 2015). Sediment accretion is a quantitative measure of water filtration function of wetlands (Gustavson and Kennedy, 2010). However, sediment accretion in wetlands is a difficult parameter to measure due to its methodological difficulties (Loaiza and Findlay, 2008).

Phosphorus (P) retention is the capacity of wetlands to remove phosphorus from the water column through physical and biological processes (Reddy et al., 1999). Phosphorus accretion process includes settling of particulate P, settling of P in biomass and precipitation of P with metal cations (Keller and Knight, 2004; Mitsch and Gossilink, 2000). Wetlands usually serve as phosphorus traps but sometimes they release phosphorus under anoxic conditions resulting in negative retention (more vegetation helps to retain P as it releases oxygen via its roots). However, the potential mobility of P in wetland sediments is still not clearly understood (Johannesson, 2008). It is observed that the amount of phosphorus retention is based on inflow- outflow, and not on the retention efficiency (% retention). The P uptake by the vegetation is retained short-term (Johannesson, 2008), whereas P in the accreted sediments is retained long-term in wetlands (Keller and Knight, 2004).

The structure of the vegetation in wetlands and the wetland hydrology play an important role in sediment accretion such as excessive waterlogging/submergence diminishes accretion rates (Delaune et al., 1994; Jarvis, 2010; Nyman et al., 2006; Stumpf, 1983; Turner, 1990). The sediment accretion rates in wetlands depend on the vegetative community, hydrologic alteration and nearness to sediment and water source (Jarvis, 2010) which are the characteristics specific to a wetland type (Table 3.1). Therefore, we used sediment accretion rate as a parameter to assess the water filtration function of each wetland type.

Development decisions are usually based on economic considerations. A major reason for huge wetland loss is that the wetland area did not compete efficiently in terms of its economic value with other land uses (Gustavson and Kennedy, 2010) and this is due to the failure of including non-market values of environmental services from wetlands in the decision making process. The valuation of wetlands becomes important in some situations e.g., when assessing the total impact of ecosystems on human wellbeing and the economy, comparing alternative development options and the impact assessment (the impact of draining wetlands on other ecosystem services) (Gleason et al., 2008). Wetlands are described as kidneys of the landscape due to chemical and hydrological processes they perform (Barbier et al., 1997). Most of the wetland services are public goods and their consumption is non-excludable. Despite being the only type of ecosystem having an environmental treaty (Ramsar) for its conservation and other legislations for protection, wetlands are being increasingly degraded due to absence of the market for their services and ignorance of policy makers and governments about their value (Ajibola, 2012). Economic valuation alone is not sufficient to avoid further degradation but broader policy interventions need to incorporate the economic values of wetlands (Newcome et al., 2005).

Table 3.1: Major wetland types and their characteristics (modified from Ducks Unlimited Canada, 2011; Kellner, 2002; Smith et al., 2007; Warner, B. G., Rubec, 1997; Zoltai and Vitt, 1995).

	Wetland type			
Attribute	Marshes	Swamps	Bogs	Fens
Definition	Shallow water areas that are mostly grasslands, can be freshwater or saltwater, amount of water in a marsh can change seasonally or with tide.	Slow moving streams, rivers or isolated low areas with more open and deeper water than marshes.	Peat lands raised or level with surrounding terrain; unaffected by runoff or groundwater from surrounding; receive water from precipitation; water table is at or slightly below surface.	Peat land with fluctuating water table at surface, water channels enter in and water seeps through peat.
Soil	Low mineral soil but substantial content of organic matter and nutrient rich.	Poorly-drained and water logged soil but nutrient rich	Low nutrient soils, peat is waterlogged, poorly oxygenated or devoid of oxygen	Solis have higher concentration of minerals than bogs and are nutrient rich.
Moisture regime	Hydric to very hydric	Hygric to hydric	Subhygric to hygric	Hygric to hydric
pH	5.2-6.4	5.9-6.1	3.5-3.6	4.0-6.2
Plant life	Freshwater marshes contain soft stemmed and non-woody plants e.g. grasses, shrubs Saltwater marshes have grasses, reeds, and rushes.	Have woody shrubs and trees rather than grasses and herbaceous vegetation	May be treed or treeless, usually covered with sphagnum spp. and shrubs which can survive in humid and nutrient poor conditions	Wetter fens are dominated by graminoid, bryophytes, sedge, rushes and moss vegetation, drier fens are dominated by trees as black spruce and shrubs
Major forms	Channel, coastal, shore, estuarine, kettle, stream, floodplain etc.	Basin, flat, spring, stream, shore, peat margin etc.	Basin, blanket, domed, flat, floating, mound etc.	Basin, channel, floating, feather, spring, stream etc.

Wetlands are complex ecosystems and their valuation is challenging. The valuation of wetlands convolutes due to unknowingness of their physical, chemical and biological characteristics and processes that generate ecosystem services. Economic valuation is sometimes based on the perceptions of benefits and preferences by the people that make it more complicated and conflictive due to the involvement of direct personal interests. To our knowledge, there is no valuation framework to value ecosystem services generated from different wetland types. The economic valuation literature for wetlands, on the other hand, is filled with misleading values which seem exaggerated and irrational; therefore, policy makers do not take these values seriously (Lambert, 2003). Other challenges include the lack of data on structure, distinction between types and neglecting the spatial significance of wetlands.

The magnitude and number of ecosystem services generated from a wetland depend on its type and ecology (Brander et al., 2006; Turner et al., 2000). Different wetland types are ecologically different (Warner and Rubec, 1997); therefore, the values of different wetland types are expected to be different for their ecosystem services. Some studies identified different wetland types on southern Ontario's landscape (e.g., Anielski and Wilson, 2010; Hotte et al., 2009), but valuated these types using a single unit value for water filtration service. The magnitude and dynamics of sediment retention depend on vegetation and elevation of the wetlands (Loaiza and Findlay, 2008) and which, in turn, are specific/unique for a particular wetland type.

Different wetland types produce an ecosystem service at a different magnitude based on their unique conditions and characteristics. Therefore, wetland types cannot be aggregated together to assign the same value for an ecosystem service. The purpose of this chapter is to demonstrate the unique functioning of each wetland type for water filtration service and translate it into economic value. Here, we present a valuation framework for wetland types and apply it to assess the value of water filtration service for sediment and phosphorus, provided by different wetland types in southern Ontario. We use the sediment accretion rate as the primary parameter to distinguish functioning of different wetland types for water filtration. To the extent of our knowledge, this is the first study that separates the biophysical/functional assessments of different wetland types to recognize their uniqueness and value for water filtration function.

3.3 Materials and methods

3.3.1 Study Area

The study area is the most southerly ecozone (mixedwood plains ecozone) in Ontario, Canada (Figure 3.1) and is country's most affected area by human activity (Taylor et al., 2014). The total area of the selected region is 5.33 million hectares which makes up 4.91% of Ontario. This region has experienced high development and agricultural pressures due to its high population (Ducks Unlimited Canada, 2010). The area is completely mapped in the Canadian Wetland Inventory (CWI) for its wetland types. Since the European settlement (c.1800), the wetlands have drastically declined by more than 70% in southern Ontario (Ducks Unlimited Canada, 2010).

The region is undergoing high economic activity and a fast growing population. Agriculture is the dominant land use and natural vegetation is reduced to three percent of its historic area. This region underwent an average wetland loss of 68% reaching to a maximum loss of 90% in some of its areas. The aquatic ecosystems are deteriorating due to sedimentation and organic pollution from intensive agriculture (Taylor et al., 2014). The wetland type area is derived from SOLRIS (Southern Ontario Land Resource Information System) land use data (MNR, 2008) (Table 3.2). The selected region contains bogs (0.85%), fens (0.58%), marshes (11.72%) and swamps (86.85%) (Table 3.2). The total area of all wetland types is 896,149 hectares which makes up 16.81% of the area of the selected region.

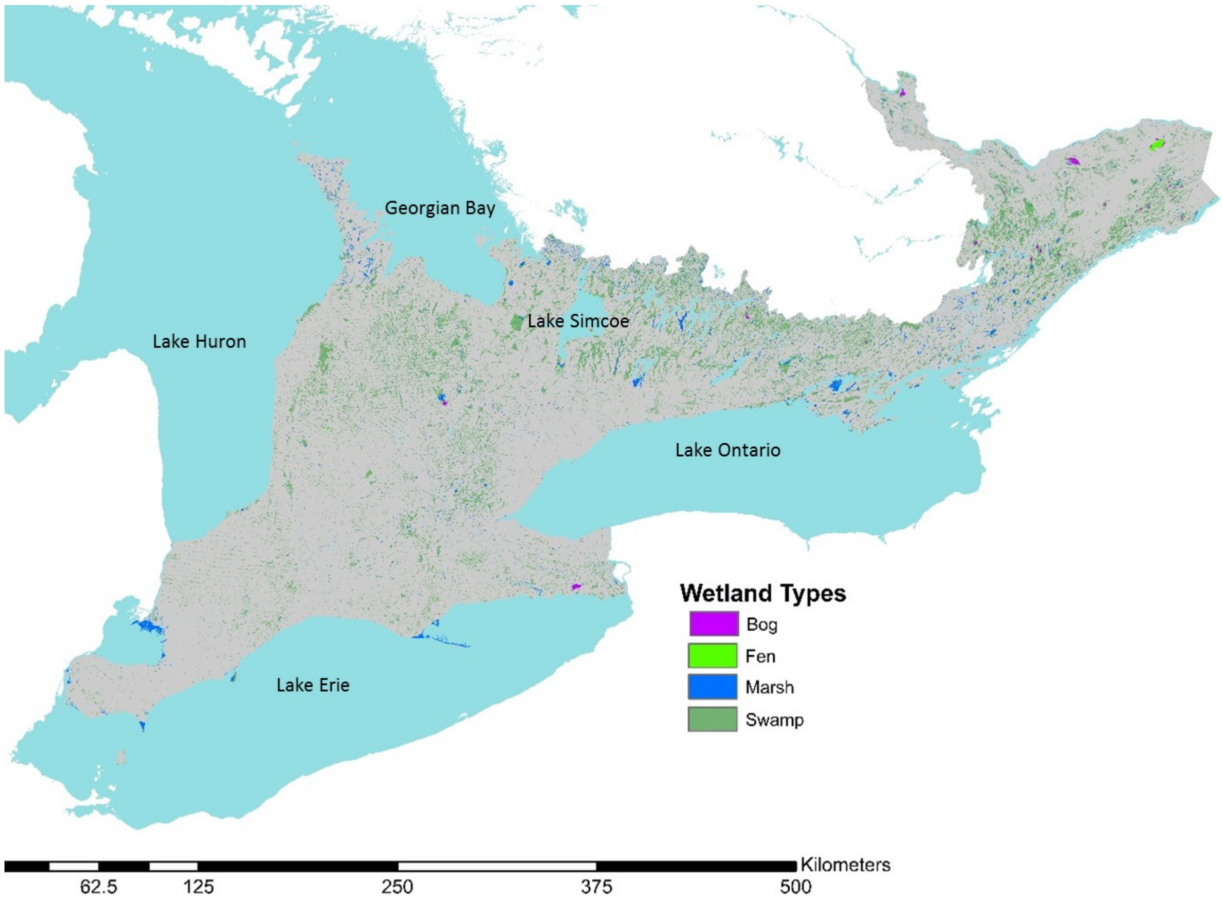

Figure 3.1: Wetland types in southern Ontario. The area in gray is the selected region (MNR, 2015).

Table 3.2: Areas (hectare) of different wetland types located in southern Ontario (MNR, 2008).

Wetland Type	**Area (ha)**	**Area (%)**
Bog	7,623	0.85
Fen	5,241	0.58
Marsh	104,991	11.72
Swamp	778,294	86.85
Total	**896,149**	**100**

3.3.2 Valuation methodology

The ecosystem services from each wetland type need separate quantification based on wetland types' characteristics and functioning. The valuation framework modified from Turner et al. (2000) is applied to determine the value of different types of wetlands. As a first step, framework suggests identification of a wetland type that performs specific physical, chemical, and biological processes. These processes are based on specific characteristics of each wetland type whose functioning results in a particular magnitude of ecosystem services. Finally, this framework integrates the (biophysical) assessment of ecosystem services with economic valuation (Figure 3.2).

The functioning of wetlands can be determined/ assessed by measuring different parameters such as sediment accretion rate, area, and biomass. The selected parameter must effectively connect the structure and functioning of a wetland with its ecosystem services. Here, we used sediment and phosphorus accretion rates to assess the water filtration service across four major wetland types. Based on literature data, we determined an average sediment and phosphorus accretion rate for each wetland type. These accretion rates link the functions and processes of each wetland type with the water filtration service and are used to generate wetland value functions (Eqs. 3.1 & 3.2).

$$V_{si} = 100 * R_i * A_i * SR_C \qquad (3.1)$$

where V_{si} is the total value ($/year) of sediment retention for ith wetland, R_i is the unit sediment accretion rate (cm/year), A_i is the area of ith wetland (ha) and SR_C is the sediment removal cost ($/m3).

$$V_{pi} = 0.1 * R_i * A_i * BD_i * Pr_i * PR_C \qquad (3.2)$$

where V_{pi} is the total value of phosphorus retention of ith wetland, BD is the bulk density of soil (Mg/m^3) for ith wetland, Pr_i is the phosphorus retention in soil (mg/Kg), and PR_C is the phosphorus removal cost ($/Kg). Finally, wetland value functions are used to estimate sediment (Eq. 3.1) and phosphorus retention value (Eq. 3.2) for each wetland type.

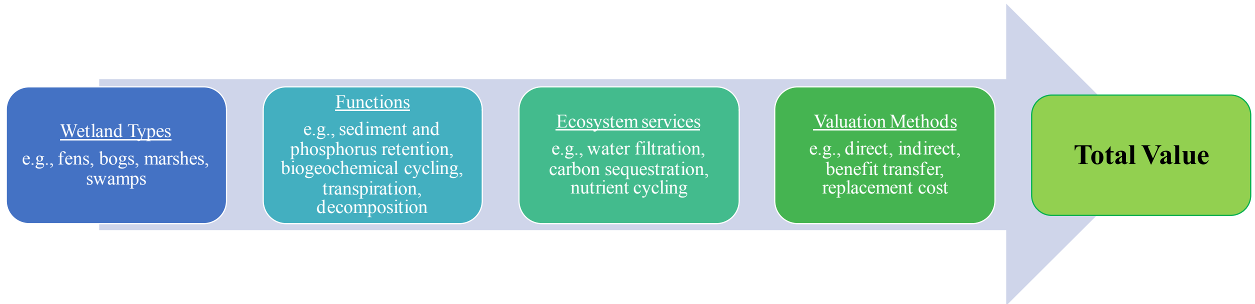

Figure 3.2: A framework for valuation of ecosystem services from different wetland types (modified from Turner et al., 2000). The framework links specificities of wetland types with the ecological and biophysical functions, which result in specific magnitude of ecosystem services.

3.3.3 Sediment accretion

We relied on literature data for sediment accretion rates in different wetland types. The two approximate methods commonly applied in literature to collect the sediment accretion data are hydrologic budget and geochemical analysis. The hydrologic budget method involves monitoring of sediment inflow and outflow from a wetland. This technique determines the sedimentation rate over a large area and results are applied to smaller units within an ecosystem. Geochemical analysis involves radiometric dating of sediment cores. This approach depends on a few samples and results of those samples are extrapolated over large area (Demissie and Fitzpatrick, 1992). In radiometric dating, radionuclides are used as chronological markers (age and depth markers). The nuclear activities and accidents released radionuclides (e.g., ^{137}Cs and ^{14}C) into the environment which are also used to date peat sequence in wetlands (Le Roux and Marshall, 2011). The two natural radionuclides that are commonly used in most of the studies are ^{210}Pb and ^{14}C (Church et al., 1987; Walker et al., 2007). The sediment accretion in wetlands is measured vertically above the marker horizon for a particular time (from time T_0 to T_1) (Figure 3.3).

3.3.4 Phosphorus retention

Phosphorus in wetland soils declines with increase in depth (Craft and Chiang, 2002; Fisher and Reddy, 2010). Total P found in surficial soils is 10 to 50% but it decreases to 5 to 10% at a depth of 60 cm (Fisher and Reddy, 2010). Wang et al. (2008) observed in four wetlands that total P becomes nearly stable at a depth of 7.5 to 10 cm and below. Therefore, we used data of the soil samples taken at a depth of 10cm and hence the P content in soils is not affected by the decomposition of trees in the wetlands (Pinder et al., 2014).

Assuming soils at equilibrium phosphorus concentration (EPC_0), we used total phosphorus content in soil samples to determine the total phosphorus retention. EPC_0 implies that sediment is neither behaving as a sink nor as a source and net P adsorption in the soil is equal to desorption (Reddy et al., 1995).

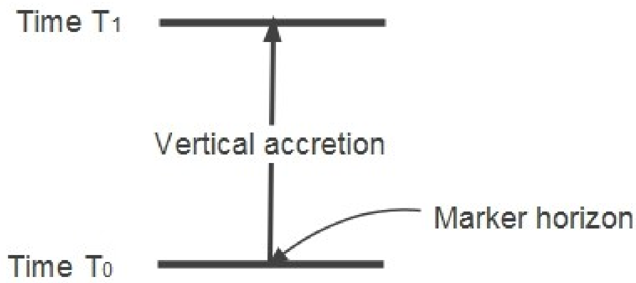

Figure 3.3: Data interpretation for accretion (or accumulation) rates from T_0 to T_1 in wetlands (modified from Batzer and Sharitz, 2006).

3.4 Results and Discussion

3.4.1 Regression models

We conducted regression analysis to investigate whether a significant relationship exists between wetland size and sediment accretion rate (Figure 3.4). The regression models for prediction of sediment retention in different wetland types are developed, where the predictor variable was wetland size (area in ha) and response variable was accretion rate (cm/year). Statistical relationships between sediment accretion rate (cm/year) and wetland size (ha) are determined by standard linear regression model for marshes and bogs. The linear regression models explain 33% of the variation in the data set for bogs and 23% for marshes. Logarithmic transformation models are fitted to the data for fens and swamps because most of the data spread. Note that sediment accretion is a key process for particle-associated pollutants, including phosphorus. The regression models show that 15% and 27% of accretion rates are predicted in the case of fens and swamps, respectively (Figure 3.4).

The low R^2 values for regression models show positive but weak correlation between wetland area and accretion rates. The positive correlation between sediment accretion rate and size (Figure 3.4) can possibly be explained by the fact that high residence times result in more sediment accretion (Kidd et al., 2015).

The sediment accretion rates are scattered for same wetland size due to changes in site-specific characteristics (e.g., inflow, outflow, surrounding land use) for each wetland type. The location and sediment entering and leaving the wetlands are the pivotal factors for sediment accretion rates in wetlands (Kadlec and Robbins, 1984). In our data set for regression models, most of the fens are less than 10 ha and bogs are less than 100 ha in area. However, marshes and swamps are usually larger than 100 ha. Similarly, marshes and swamps are the only wetland types that exceed 100 ha in size in the selected region of southern Ontario.

The measurement methods also affect the rate of sediment accretion. For example, short term measurements (using feldspar marker) can give higher rate of sediment accretion compared to long term measurements (using ^{137}Cs and ^{210}Pb) because short term measurements do not account for shallow subsidence within top layer of sediment (Ensign et al., 2014). In our analysis, accretion rates (Figure 3.4) are mostly taken from the studies which use long term measurement techniques (e.g., Church et al., 1987; Craft, 2007; Neubauer et al., 2002).

Ghermandi et al. (2007) developed a meta-regression model for the value of ecosystem services (e.g., water quality amelioration, recreation, biodiversity and natural habitat provision) from natural and

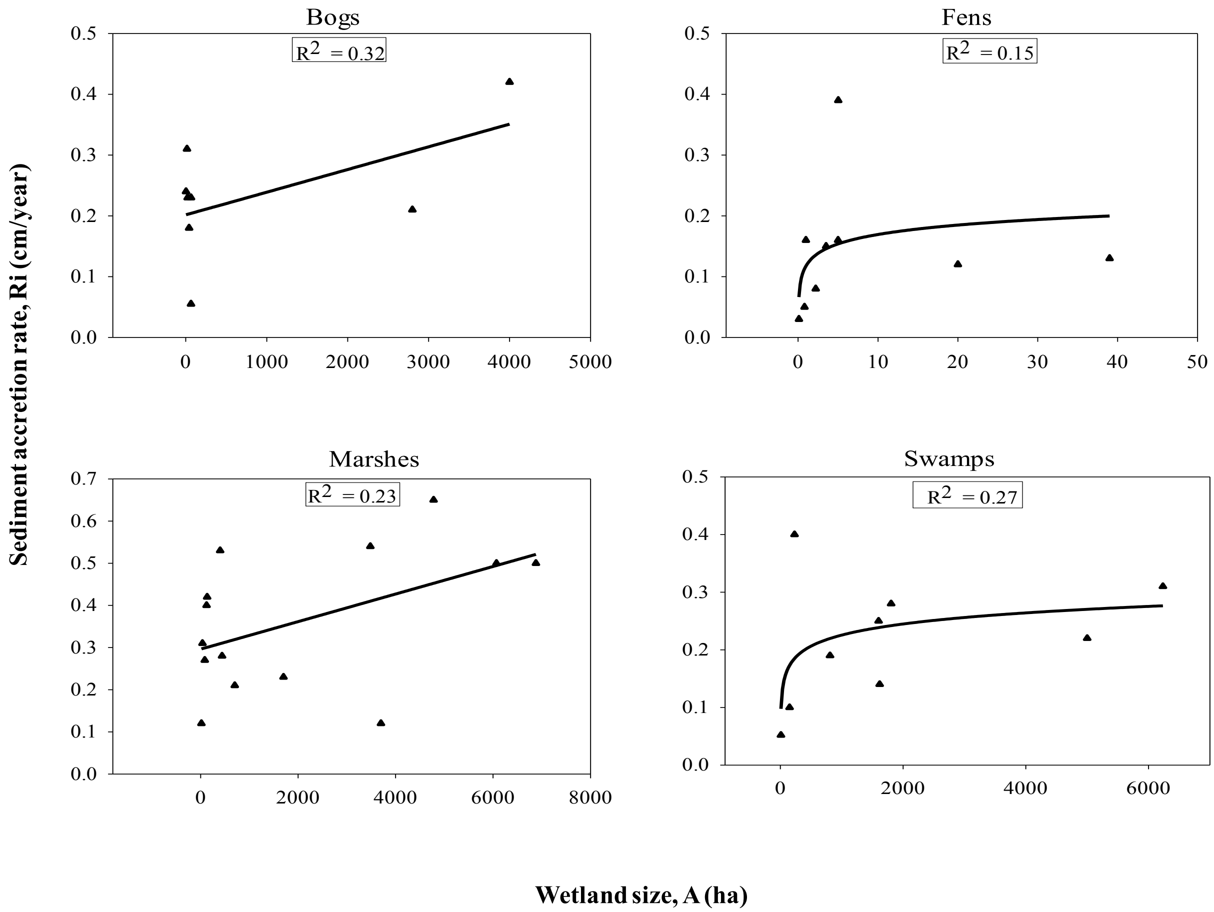

Figure 3.4: Regression models for different wetland types between sediment accretion rate (cm/year) and wetland size (ha).

constructed wetlands, and showed negative correlation between the unit value ($/ha/year) and the area of wetlands. Brander et al. (2006) used meta-analysis of literature which substantiates that there is no obvious relationship between the unit value of a wetland and its area. Both of these studies perform the analysis for a unit value of a suite of ecosystem services generated from wetlands. We, however, use a single service or function in our work. It is important to point out that the value of certain/specific services (e.g., hunting) may have a stronger relationships with the size of wetlands than others.

3.4.2 Sediment retention

For all wetlands types, there is a positive sediment accretion rate and wetland area (Figure 3.4). The correlations are weak, however. In particular in small wetlands (< 1000 ha), the reported accretion rates tend to be highly variable with little discernible trends. As a result, some smaller wetlands have higher sediment accretion rates than their larger counterparts (Table AB1-AB4). Clearly, environmental factors other than overall size influence sediment accumulation in wetlands. Pending a more detailed analysis, in what follows we used the average accretion rate for each wetland type to estimate the annual sediment retention in bogs, fens, marshes and swamps, while relying on the standard deviation as a measure of the variability (Table 3.3). The sediment accretion rates are highest for marshes and lowest for fens among. The increasing order of sediment accretion rate for wetland types is listed as: marsh>bog>swamp>fen.

3.4.3 Phosphorus retention

Fennessy et al. (2004) dataset, from the samples collected at a depth of 10 cm, is used to draw the P content in wetland soils. The average P content found in soil samples of swamps (forests and shrubs), marshes (depressional, mainstream and headwater), fens (meadows and calcareous) and bogs is 765±195, 910±315, 795±250 and 1070 mg/Kg (or ppm), respectively (Fennessy et al., 2004). Another study on 15 marshes within the Painter Creek Watershed in Minnesota (USA), found that total phosphorus (±SD) in the upper 0-30 cm of soil was in the range of 1160±320 ppm (mg/Kg) (Bruland and Richardson, 2006).

Based on measurements of wetlands in Ontario and Alberta, the soil densities for bogs, fens, marshes and swamps are 1.49, 1.54, 2.0 and 1.57 Mg/m^3 respectively (Redding and Devito, 2005; Thomas and Sevean, 1985). However, to keep it consistent with P content (in soils) data used in our analysis, we used the average density of 1.75 g/cm^3 or Mg/m^3 for wetlands reported by Fennessy et al., 2004 for calculation of total P retention in wetland types (Fennessy et al., 2004) (Table 3.3).

Table 3.3: Sediment and P retention rates (m^3/ha/year) based on sediment accretion rates (cm/year) in different wetland types.

Wetland Type	Sediment accretion (cm/year)	Sediment retention rate (m^3/ha/year)	P content in soil (mg/Kg)	P retention rate (Kg/ha/year)
Bog	0.23±0.1	23±10	1070[1]	43.1±18.7
Fen	0.14±0.1	14±10	795±250[1]	19.5±15.2
Marsh	0.36±0.2	36±20	1035±225[1,2]	62.5±38.9
Swamp	0.22±0.1	22±10	765±195[1]	29.5±15.4

[1](Fennessy et al., 2004); [2](Bruland and Richardson, 2006)

The equilibrium phosphorus concentration (EPC_0) is considered for sediment retention where net adsorption and desorption of phosphorus are equal from soil sediments. However, there can be phosphorus release than retention in wetlands with lower phosphorus loading (Bostrom et al., 1982; Froelich, 1988; Logan, 1982; Reddy et al., 1995). Therefore, we considered only long term sustainable P retention process in sediment accretion which can only vary depending on impaction. The results reported in different studies showed that P accretion rate could vary from 0.4 to 4 g P/m^2/year (4 to 40 Kg P/ha/year) in wetlands (Craft and Richardson, 1993; Dunne and Reddy, 2005) which is identical to the range of phosphorus retention (Table 3.3) estimated in our analysis. The other similar studies conducted in North Dakota, Georgia and Florida (US) showed that the storage of phosphorus in wetlands ranges from 0.1 to 50 Kg P/ha/year (Craft and Casey, 2000; Dunne et al., 2006; Freeland et al., 1999; Marton et al., 2015).

The phosphorus retention rates reported in the literature for constructed wetlands fall in the range of 1-58 kg/ha/year (Johannesson et al., 2011) which are similar to the range of P retention rates used in our analysis. The study conducted on Old Woman Creek wetland (marsh) in the western basin of Lake Erie estimated that there is P retention of 50-70 Kg P/ha/year. The study also concluded that the restoration of one-fourth of the original wetland area could reduce 25 to 30% of phosphorus loading to western Lake Erie (Mitsch et al., 1989; Shane et al., 2001). Another mass balance study of phosphorus retention on Hidden Valley wetland, Ontario, found that there is 50% of total phosphorus retention in the wetland, but the plant available phosphorus (orthophosphorus) exports were 22% more than the imports (Gehrels and Mulamoottil, 1990; Shane et al., 2001). Johannesson et al. (2011) compared phosphorus retention in a constructed wetland measured by hydrologic budget method (inflow-outflow) with the calculations of phosphorus accumulated in the sediments. The results showed that the phosphorus retention is 17% of the P load by inflow-outflow measurements and 80% of the P load calculated from sediment accretion for four years. The study explained that the low phosphorus retention through inflow-outflow measurements is due to underestimation of P load at the main inlet of the wetland. However, the higher P retention (80%) calculated by sediments accretion is similar to the P retention in many other studies of the constructed wetlands (e.g., Braskerud et al., 2000; Carleton et al., 2001). Therefore, study emphasized the need of further analysis for better understanding the discrepancy of P retention in inflow-outflow measurements and sediment accretion calculations (Johannesson et al., 2011).

Phosphorus retention rates are not only dependent on concentration of P in wetland soils, but also on the sediment accretion rates. For example, fens have higher phosphorus concentration in the soils but less sediment accretion rates compared to swamps. This fact results in lower phosphorus retention rates for fens than that of swamps (Table 3.3). The freshwater wetlands receiving no anthropogenic loads were reported to have phosphorus retention rates of 1.7 to 7.3 Kg P/ha/year (Johnston, 1991). One possible explanation of this P load could be atmospheric P deposition as in the case of Lake Erie where 6% of the total P load comes from atmospheric deposition (Boehme et al., 2013). Phosphorus accumulation in wetlands was found as 5.7 Kg/ha/year in agricultural watersheds and 3 Kg/ha/year in non-agricultural watersheds in Prairie Pothole region of South Dakota (Johnston, 1991; Riemersma et al., 2006). These studies about phosphorus accumulation show that it is highly variable across landscapes. Therefore, in an effort to best capture the variation in the values, high standard deviation in the mean P retention rates were introduced in our analysis.

3.4.4 Wetland value functions (V_{si} , V_{pi})

The average sediment removal cost (SR_C) from 10 storm water management facilities (SWMFs) estimated at \$170±78/m^3 (Aziz et al., in prep.) is applied to determine the unit values (Table 3.4). The total phosphorus removal cost (PR_C) based on historic performance of 12 Water Pollution Control Plants (WPCP), a wastewater treatment center (WWTC) and a sewage treatment plant (STP) is estimated at \$19±13/Kg TP in CAD 2016 (Aziz et al., in prep.). The unit values of different wetland types for phosphorus retention are obtained by multiplying phosphorus costs with their phosphorus retention rates (Table 3.4).

The unit values for phosphorus retention follow the same order of increase as the unit values of sediment retention for wetland types (Figure 3.5). It shows that the P retention is strongly correlated to sediment retention.

The estimated unit values (\$/ha/year) for water filtration service provided by bogs, fens, marshes and swamps are 4730±2560, 2750±2055, 7310±4545and 4300 ±2460, respectively. According to our results, marshes are the most valuable wetlands for water filtration. However, higher value of a wetland type for one ecosystem service does not imply a higher value for other ecosystem services. A study on the Lake Simcoe basin's natural capital recognized the different types in the basin but used the same value of \$466/ha/year (CAD, 2016) for their water filtration service. This value is deduced from a statistical analysis of potential increase in water treatment costs due to reduction in wetland cover in US (Wilson, 2008a). Additionally, Anielski and Wilson (2009) also identified different wetland types in the land use data but applied the same unit value of \$452/ha/year (CAD 2016 using inflation calculator of Bank of Canada) for water filtration service based on meta analysis for freshwater wetlands.

Table 3.4: Unit values for sediment retention (Vsi) ($/ha/year) and phosphorus retention (Vpi) by different wetland types using wetland value functions (Eqs. 3.1 & 3.2). The bulk density of soil used to compute P retention is 1.75 Mg/m^3.

Wetland Type	Sediment accretion rate, R_i (cm/year)	Phosphorus content, P_{ri} (mg/Kg)	Sediment retention value, V_{si} ($/ha/year)	P retention value, V_{pi} ($/ha/year)
Bog	0.23±0.1	1070[1]	3910±2470	820±665
Fen	0.14±0.1	795±250[1]	2380±2020	370±385
Marsh	0.36±0.2	1035±225[1,2]	6120±4410	1190±1100
Swamp	0.22±0.1	765±195[1]	3740±2415	560±480

[1](Fennessy et al., 2004); [2](Bruland and Richardson, 2006)

Another study of ecosystem services on Greenbelt, Ontario (Wilson, 2008b) also assigned a single unit value of \$566/ha/year (CAD, 2016) to different wetland types for water filtration service. To derive this unit value, they used avoided cost method based on potential increase in water treatment costs due to decrease in forest cover. Conversely, we estimated a higher (about fivefold) unit value of water filtration service for each wetlands type based on their capacity to retain sediment and phosphorus (Table 3.4). At global scale, wetlands are assigned a value of \$259/ha/year (CAD 2016) for water filtration (Schuyt and Brander, 2004), which is a lower value compared to our estimates (Table 3.4). Furthermore, Schuyt and Brander (2004) assign a unit value of \$ 8/ha/year (CAD 2016) on the freshwater marshes for all of their ecosystem services in North America, while our analysis computed a unit value of \$650 - \$14000/ha/year for one ecosystem service (of water filtration for sediment and phosphorus). Therefore, this mismatch of results suggests that the value of water filtration service from wetlands is underestimated at global scale and needs a reconsideration/review. This high variability in the values may be due to the use of contingent valuation methods and providing of limited information to respondents by most of the studies in literature (Ghermandi, 2005). However, several studies (e.g., Breaux et al., 1995; Lambert, 2003) already suggested the use of cost-based approaches for valuation of water quality service from wetlands to match the original benefits.

3.4.5 Total value of wetland types

The unit values for water filtration service by each wetland type are applied to the respective wetland area in southern Ontario to obtain the total values of phosphorus and sediment retention by all wetlands in the region (Table 3.5). Even though the unit value for swamps is approximately half of that of the marshes for water filtration service, the total value of swamps in southern Ontario (Figure 3.6) is highest because of the fact that they make up the most (86.85%) of the total wetland area in the region.

In this analysis, we generalized some parameters at the regional level such as sediment and phosphorus retention for a specific types of wetlands. However, a site specific study can provide more accurate results because different characteristics and functions of a wetland (e.g. inflow, outflow, vegetation type, area, and accretion rate) can be measured accurately at the respective site. The total value of water filtration service (for sediment and P removal) performed by all wetland types in southern Ontario is \$4.58±1.98 billion per year (CAD 2016). The value of sediment retention in wetlands is 6.25 times higher than the value of phosphorus retention.

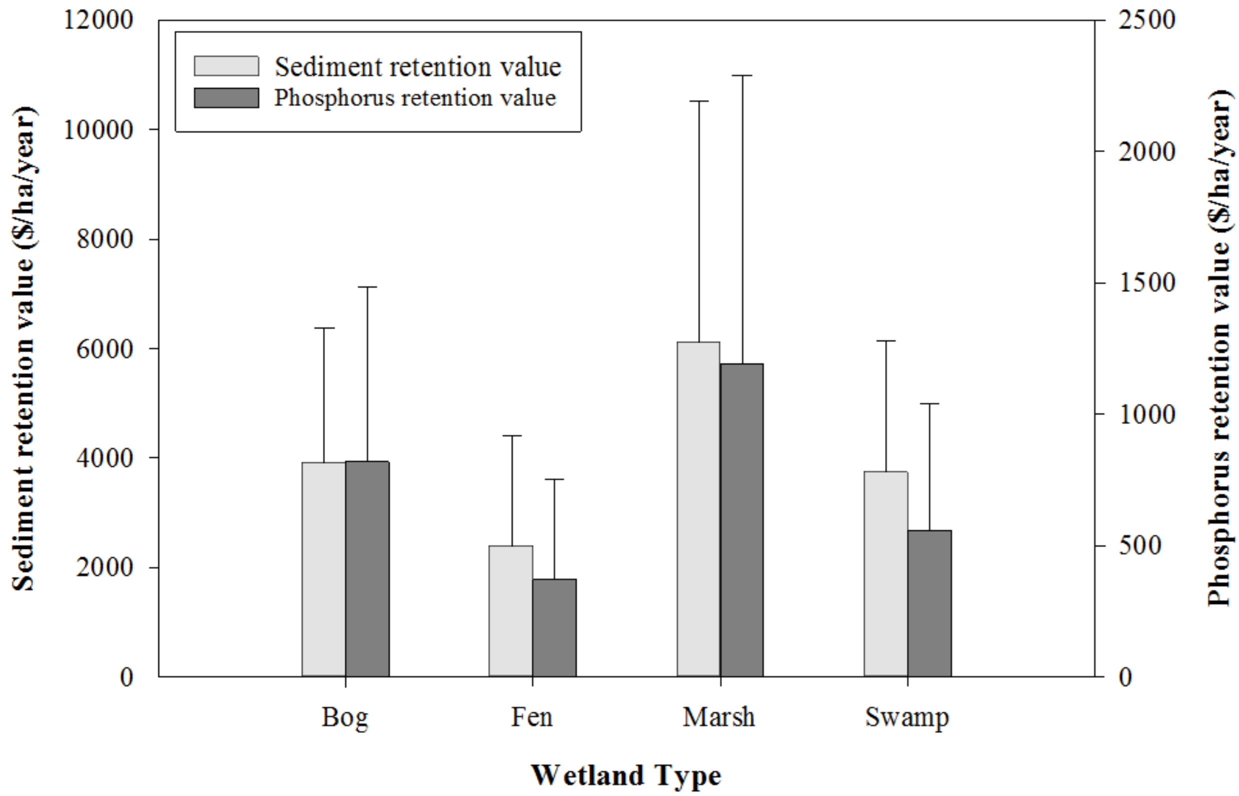

Figure 3.5: Unit values ($/ha/year) of sediment and P retention in different wetland types located in southern Ontario (error bars show the standard deviation in the mean values).

Table 3.5: Values of sediment retention, P retention and total (sediment+P retention) for differentwetland types in southern Ontario.

Wetland Type	**Area (ha)**	**Sediment retention value (10^6×$/year)**	**Phosphorus retention value (10^6× $/year)**	**Total value (10^6×$/year)**
Bog	7,600	30±19	6±5	36±19
Fen	5,240	13±11	2±2	14±11
Marsh	105,000	645±465	125±115	770±475
Swamp	778,300	2,910±1880	435±373	3,345±1915
Total	**896,140**	**3598±1936**	**569±390**	**4165±1975**

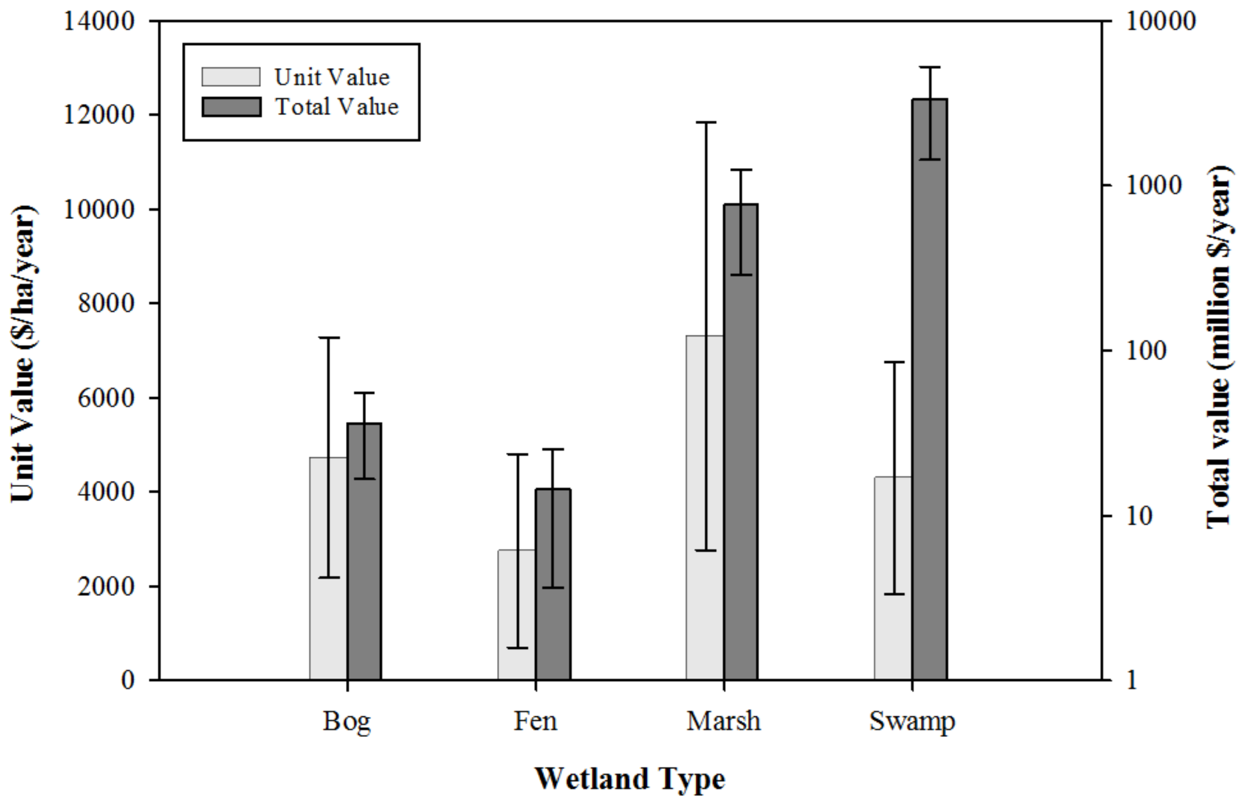

Figure 3.6: The unit and total values of wetland types located in southern Ontario for water filtration service (for sediment and phosphorus removal) (error bars show standard deviation in the mean value).

3.4.6 Cost-effectiveness analysis of different options to offset P lost from conversion of existing wetlands to agriculture

Phosphorus is the ultimate limiting nutrient in freshwaters and lakes in and around southern Ontario (Ontario Ministry of the Environment, 2012; Schindler, 2012). The only method that had proven success in controlling eutrophication and recovering lakes located in this region (e.g., Lake Erie, Lake Ontario) is to reduce P inputs (Schindler, 2012).We assessed the cost to offset lost P under a scenario where all existing wetlands are converted to agriculture/farming. The total P retention in wetlands was estimated as 29,952± 11,985t P/year based on the unit retention rate (Table 3.3) and the area of the wetlands. The additional load from converting wetland areas to agriculture was calculated using the average P delivery rate for row crops, small grains, forage and pasture from the local and regional studies (Donahue, 2013; Hore et al., 1973; Jeje, 2006; Shaver et al., 1994; Winter, 1998). Using the average delivery rate of 0.52±0.28 Kg ha^{-1} $year^{-1}$, the additional P load was calculated as 465,998±250,920 Kg P/year or 466±251 t P/year. Therefore, the total phosphorus load (P lost+ additional) from wetland loss and additional P resulting from the conversion of wetlands to farming was 30,418,360± 11,988,355 Kg P/year or 30,420±11,990 t P/year. We used three alternatives to derive the cost to offset this P load: 1) best management practices (BMPs), 2) constructed wetlands (CWs), and 3) wastewater treatment plant (WWTP) upgrades. The cost of lost P from converting a hectare of a wetland type and converting all wetlands to agriculture is estimated using these three alternatives (Table 3.5).

3.4.6.1 Via BMPs

The South Nation Conservation (SNC), Ontario, derived the cost of $400/year (CAD 2009) for removing one Kg of P from delivered/completed BMPs projects in the watershed. This includes the cost of BMP construction and project management which is applied to the P discharger. This figure is accepted by all stakeholders (Ministry of Environment MOE; farmers, and wastewater treatment plants). Water quality trading is a tool to improve water quality in Ontario which is already implemented by the South Nation River and the Nottawasaga Valley conservation authorities and is currently being considered in the Lake Simcoe watershed (Marcano, 2015). SNC revises this figure each year for annual inflation. We revised it for 2016 and used $ 447 per Kg of P removal per year (CAD, 2016) to calculate the cost to offset phosphorus in BMPs. The final cost of P retention via BMPs is obtained (Table 3.6) by applying this value to the total P loss from conversion of wetlands to agriculture.

3.4.6.2 Via constructed wetlands (CWs)

In order to determine the cost of offsetting P, it is crucial to identify the P retention rates in newly constructed wetlands. Kynkäänniemi et al. (2013) showed that the newly constructed wetlands can retain 69±36 Kg/ha/year of total phosphorus TP (based on two years of operation) (Kynkäänniemi et al., 2013). Using this retention rate and total P released from the conversion of wetlands to farming, the area of the constructed wetlands required to offset P load is 440,846±288,255 ha, which is 50% of the existing wetland area. The annual cost of a functional wetland of size 1.125 ha, operating in (Embrun) eastern Ontario, with an estimated lifespan of 30 years, is \$ 5220 (CAD1997) or \$ 7420 (CAD 2016). The cost is based on interest on capital investment, operation and maintenance cost, annual depreciation and loss of crop yield on the land (Tousignant et al., 1999). We derived the unit cost of \$6,596/ha/year (CAD 2016) for our estimation (Table 3.5).

3.4.6.3 Via WWTPs upgrades

For this scenario, the upgrading of existing wastewater treatment plants is considered to counterbalance the loss of wetlands. A report on cost benefit analysis of phosphorus in the Grand River watershed, Ontario, showed that if all the WWTPs are upgraded in the watershed, it will cost \$5147 to remove one Kg of P (CAD 2011) (Hanna, 2015b) or \$5475/Kg of P (CAD 2016). This cost does not include the optimization of operation of current processes in the upgrading option. Using this cost, WWTPs become the most expensive option to offset the lost P from conversion of wetlands to agriculture (Table 3.6).

This analysis (Table 3.6) shows that any option of phosphorus removal will be counterproductive in the absence of wetlands and it will be highly expensive to offset phosphorus load resulting from conversion of existing wetlands to agriculture. The least expensive option is constructed wetlands, which will cost \$2.9±1.9 billion per year. The areas required for constructed wetlands to counteract the P load from the loss of each hectare of bog, fen, marsh and swamp are 0.62, 0.28, 0.41 and 0.91 ha respectively. The required area is almost equal in the case of marshes (0.91 ha; because their P retention rates are equal) but is less for all other wetland types because of their comparatively low P retention rates. In addition to area requirement, stakeholders have to bear the annual cost (Table 3.6) for loss of each hectare of a wetland. All these considerations make constructed wetlands an unviable option.

Table 3. 6: Cost of different alternatives to offset phosphorus released from conversion of existing wetlands to cultivated lands (all values are in CAD 2016).

Alternatives	**Cost (×10^3) \$ per year to offset lost P from conversion of one hectare of a wetland**				**Cost (billion \$/year) to offset P when all wetlands are converted**
	Bog	Fen	Marsh	Swamp	
BMPs	19.2±8.4	8.7±6.8	27.9±17.4	13.2±6.9	**13.60±5.35**
CWs	4.1±2.8	1.9±1.7	6.0±4.9	2.8±2.1	**2.90±1.90**
WWTPs	236.0±102.4	106.8±83.2	342.2±213.0	1611.5±84.3	**166.50±65.60**

3.4.7 Limitations

There are a number of limitations in this study that need to be discussed. For example, sediment retention depends on the sediment density which varies from location to location. This factor is not incorporated and only sediment accretion rates are taken into account. Sediment retention and accretion rates also depend on the land use upstream of the wetland and wetland location in the landscape. Because these parameters are site specific, they are not considered in this study.

Additionally, wetlands located downstream in a watershed have greater effect on water quality than those located upstream; therefore, this spatial heterogeneity (or dimension), another location specific characteristic, is also excluded in our analysis. The accretion rates vary when measured with different techniques e.g., using dendrogeomerphic technique (tree-ring), sediment pins and elevation surveys for dating (Kidd et al., 2015). Sediments and nutrients accumulation depends on watershed size, land use and connectivity of wetlands to rivers and streams (Craft and Casey, 2000). This may be one reason for high variation in sediment accretion rates in wetlands.

The above limitations can be addressed by undertaking site specific analysis of wetland types because most of the functions such as sediment and phosphorus removal in wetlands are strongly related with the loads entering and leaving the wetland (Son et al., 2010).

3.5 Conclusions

Wetland types are valuated for water filtration service based on their sediment and phosphorus retention function. We used the average sediment accretion rate as the key water filtration parameter. The unit values ($/ha/year) of water filtration service, for four wetland types in southern Ontario, increase in the order of marsh > bog > swamp > fen. The main findings of this study are summarized as follows:

- The regression models showed positive but weak correlation between wetland size and sediment accretion rates.
- Sediment and phosphorus retention rates vary between different wetland types.
- The unit value of water filtration service by a wetland depends on its capacity to retain sediment and phosphorus; therefore, each wetland type has a different unit value.
- Marshes are the most valuable wetland type for water filtration because of their highest unit value among the wetland types.
- It is very costly to replace wetlands' water filtration service with human-made infrastructure.

Chapter 4

Critical assessment of the value transfer method: unit values and land use resolution

4.1 Summary

Economic valuations of ecosystem services are often based on the value transfer method where global unit values are applied worldwide. While this approach produces quick results, its reliability depends on how representative the global unit values are within a given regional context. Here, we compare unit values from two commonly used global databases, that of Costanza et al. (1997) and the Ecosystem Services Valuation Database (ESVD) (2012), to those of a regional database derived from five watershed studies. The global unit values for terrestrial biomes are generally significantly higher than the regional values. We further compare the predicted monetary values of selected ecosystem services (water filtration, nutrient cycling and carbon sequestration) in the Grand River watershed in southern Ontario, Canada, for which primary (local) unit values are available. Assuming that the local unit values are the most accurate, the regional database provides a much closer agreement than the global databases. In particular for the water filtration service, ESVD predicts a much higher value than either the regional or local estimates. The valuation results also reflect the degree of aggregation of land use categories: the explicit representation of sub-categories of forest and agricultural land increases the combined values of the three ecosystem services by 12 and 2%, respectively. Overall, our results emphasize the need to establish regional datasets to improve the application of the value transfer method.

4.2 Introduction

The science of economic valuation of ecosystem services has made significant progress, yet many challenges remain to implement its outcomes at national and regional levels (Polasky et al., 2015; Small et al., 2017). The value transfer method — a quick and popular method to assess the value of ecosystem services — uses secondary data, generated applying other economic valuation methods (Rusche et al., 2013), which reduce its accuracy. In addition, a wide variety of valuation methods used to value ecosystem services further reduces the accuracy of the benefit transfer method.

The valuation methods that use primary data, can be broadly categorized into three types: direct valuation-, revealed preference-, and stated preference- methods (Barton et al., 2015; European Commission, 2008; Rusche et al., 2013). The first type of methods, the direct valuation methods, are the market price- and cost-based- methods such as replacement cost, avoided damage cost, mitigation cost, substitute cost, production function, etc. (Barton et al., 2015). Often the market price method is used for assessing market values whereas cost-based methods are used for the non-market values of ecosystem services (Brown et al., 2007). At the same time, the cost-based methods are more meaningful and a direct way to assess the value of non-market services (Sundberg, 2004). Further, cost-based approaches are used to derive the value of ecosystem services based on the cost of replacing or mitigating the services (Swinton et al., 2007) and are useful in the cases where physical processes are well understood (Green, 2002). Since we do not know the market value of non-market ecosystem services, these methods become more effective in calculating values if a perfect substitute is present with similar benefit provision as the original system. Moreover, these cost-based methods assign values based on markets (Wyatt, 2009). However, it is impossible to value cultural services using cost-based approaches (Ledoux and Turner, 2002).

The second type of methods, the revealed preference approach, uses market prices and observations to put a price on non-market goods and ecosystem services. This approach depends on individuals' demand for an ecosystem service and employs different techniques for valuation such as travel cost, hedonic pricing method, market price, avoided cost methods, etc. (Kennedy, 2014; Rusche et al., 2013). Finally, in the third type of methods, the stated preference methods, individuals state their preferences from a set of options (Kroes and Sheldon, 1988). These are survey-based methods involving a questionnaire and hypothetical payment scenarios. The stated preference methods include contingent valuation and choice modeling methods (Rusche et al., 2013).

In this paper, we test the effect of different datasets (unit values and land use) on the value of ecosystem services in a watershed using the value transfer method. The value transfer method is a classical technique used for crude approximation of economic value of ecosystem services in the river watersheds. This is not a specific method but uses available information from sundry methods (discussed above) to valuate the region with various ecosystem types. The value transfer method is a fast, relatively inexpensive, commonly used technique for valuation of watershed ecosystem services. It is also the most popular method in the settings lacking system- (ecosystem-) specific information (Pascual et al., 2010). River watersheds are mosaics of ecosystems and value assessment of their ecosystem services is not possible using a single approach (Lee et al., 2010). Similarly, watersheds contain varied ecosystem types and therefore their primary valuation becomes prohibitively expensive (Costanza et al., 2006). Thus, the application of the value transfer approach to watersheds needs to adopt unit values from a number of studies developed using a variety of valuation methods such as stated preference, revealed preference, and cost-based methods. This disparity of methods and heterogeneity of the studies increase the uncertainty in ecosystem services value and, hence, cast doubt on their validity (Kennedy, 2014).

The value assessments based on the value transfer method vary in comparison with other methods that need primary data. Kennedy (2014) compared the value transfer and cost-based approaches by valuing a region and showed that the value transfer method imputed a (3 times) higher value for ecosystem services than the cost-based approach. However, the use of the value transfer method becomes inevitable in data-scarce regions and under certain restrictions (on budget and time). These hindrances result in accruing the benefits of local ecosystem services at regional and global scales (RS de Groot et al., 2010; Pandeya et al., 2016).

Recent advancements in valuation science are constantly developing new data on ecosystem services at the local scale (Pandeya et al., 2016). But there are still many regions/areas with missing local data; valuation studies on these regions use global unit values (using value transfer method) for economic valuation of ecosystem services (e.g., Tolessa et al., 2016). Therefore, a more meaningful approach is required for such regions with scarce data. To develop an integrated approach for ecosystem services, it is important to assess the weaknesses of existing valuation approaches (Pandeya et al., 2016). Current valuation approaches are piecemeal, unconnected, and therefore do not meet policy needs. Different valuation approaches lead to different values of the same ecosystem type and for the same service as well. Subsequently, comparison of economic values derived using different techniques is difficult at national and regional levels (Schuhmann and Mahon, 2015). Even the same valuation approach (e.g., value transfer) based on different datasets (unit values) may result in different values at a watershed scale. In fact, at the watershed scale, comparison of economic values based on different datasets remains largely unexplored. To bridge this gap, we used different datasets for a watershed valuation to show these differences and seek out any common ground among these datasets.

The cost-based approach requires three conditions to be satisfied: 1) perfect substitute, means that human engineered system must provide an equivalent function in quality and quantity to ecosystem services; 2) least cost alternative means that cheapest human engineered system should be used as a replacement for an ecosystem service; and 3) individuals must be willing to incur the cost in the case of ecosystem service loss (Sundberg, 2004). Most of the studies consider close substitutes as the approximate perfect substitute (Sundberg, 2004).

The value transfer is a classical method that uses unit values ($/hectare/year) for valuation of ecosystem services and is being extensively used for valuation of different regions worldwide (Table 4.1). Despite

Table 4.1: **V**aluation studies from around the world based on value transfer method that used unit values from Costanza et al. (1997) and ecosystem services valuation database (ESVD) (2012) datasets to valuate ecosystem services.

Study Area	**Reference**	**Dataset used**
Chongming Island, China	(Zhao et al., 2004)	Costanza et al. (1997)
Central highlands, Ethiopia	(Tolessa et al., 2016)	Costanza et al. (1997)
Changsha, China	(Yun-guo et al., 2009)	Costanza et al. (1997)
Texas, USA	(Kreuter et al., 2001)	Costanza et al. (1997)
Sanjiang Plain, Northeast China	(Wang et al., 2006)	Costanza et al. (1997)
Wenzhou, China	(Tong et al., 2007)	Costanza et al. (1997)
Shenzhen, China	(Tianhong et al., 2010)	Costanza et al. (1997)
Portugal	(Lopes et al., 2015)	ESVD (2012)
China	(Li et al., 2016)	ESVD (2012)
Taiwan	(Yuan et al., 2017)	ESVD (2012)
Okanagan, Canada	(Parrott and Kyle, 2014)	ESVD (2012)
Bhutan	(Kubiszewski et al., 2013)	ESVD (2012)
Czech Republic	(Frélichová et al., 2014)	ESVD (2012)
Asia	(Brooks et al., 2014)	ESVD (2012)

benefit transfer being a quick and less expensive method, the question is how valid is it compared to the other non-market valuation approaches outside the realm of benefit transfer. Accordingly, unit values are the essential component for accuracy and reliability of monetary estimates based on value transfer method (Wang et al., 2014).

The flow of ecosystem services, and thus their value, is strongly affected by land use change (Fürst et al., 2013; Palomo et al., 2014; Si et al., 2014; Su and Fu, 2013; Vitousek et al., 1997; Yang et al., 2009). Improvement in health and sustainability of ecosystems hinges on the relationship between value of ecosystem services and land use change (Fang et al., 2014). Ecosystem services assessment is an effective and attractive tool to inform and support land use decisions between different land use options (Förster et al., 2015). The change in the land use brings huge changes to the value of ecosystem service. Hence, land use data have been widely used in studies of environmental and ecosystem changes, and natural resources management.

With advancements in remote sensing, the high spatial resolution land use datasets are available at local and regional scales (Chen, 2012). It is very likely for the studies based on global datasets (Table 4.1) that there may be a mismatch between land use data and the global biomes for which unit values are derived, as in the case of Tolessa et al. (2016). Sometimes, these unit values are available for coarse land use resolution, therefore their application on high resolution land use data needs aggregation of subcategories (e.g., deciduous, mixed forests) into a major category (e.g., forest). Therefore, we explored the effect of resolution of land use data on the value of watershed for its ecosystem services. We attempted to quantify the difference in the values of a watershed using local and global unit values. The values of watershed ecosystem services based on global unit values using benefit transfer approach and local unit values based on cost-based approach are compared. This analysis becomes particularly important in the case of informing land use decisions based on the value of ecosystem services.

To our knowledge, the difference in the value of ecosystem services in a watershed based on different datasets of unit values and land use resolutions has been scarcely investigated. Therefore, in this study we explore the difference in the values of ecosystem services in a watershed which were obtained using datasets of local, regional and global unit values, and land use data of two different spatial resolutions.

In specific, we investigate two major research questions:

1- How reliable are the global unit-value datasets in relation to local and regional dataset?
2- What is the impact of low (coarser) and high (finer) resolution land use data on the value of ecosystem services in a watershed?

To answer these questions, we use four datasets including two global, a regional and a local dataset. Finally, we test the effect of each spatial resolution of land use data on the value of biomes by applying the local unit-value dataset.

4.3 Materials and Methods

4.3.1 Study Area

We evaluated the impact of different datasets and land use resolution on the Grand River watershed. It is the largest watershed in southern Ontario, Canada, that covers an area of ~ 680,000 hectares and flows into Lake Erie at Port Maitland (GRCA, 2015). It is a multi-use watershed with agriculture as dominant (66%) land use. It has a population of about 1 million with Kitchener-Waterloo as the largest urban area in the watershed. Olewiler (2004) assigned a highest unit value ($/ha/year) to Grand River watershed for its natural capital and ecosystem services, compared to Upper Assiniboine and Mill River watersheds—both located in Prince Edward Island, Canada. Increasing urban and agricultural pressures are threatening ecosystem services and other creatures living in natural areas in the Grand River watershed (Brox et al., 1996; Olewiler, 2004). The most recent available land use data for the watershed is for year 2015 which is obtained by merging updated wetland cover with year 1999 land use in the watershed (Table 4.2).

4.3.2 Comparison of datasets and land use resolution

Different datasets will be compared for the value of watershed using value transfer method. There are two global datasets on unit values being commonly used for valuation of ecosystem services in different parts of the world (Table 4.2) and we, here, referred them as Global 1 and Global 2. Global 1 is the Costanza et al. (1997) dataset and global 2 is the ESVD (2012) (de Groot et al., 2012) dataset. The regional and local datasets are developed for comparison and all the values are standardized to CAD 2017 using inflation calculator of Bank of Canada.

The total value of watershed based on each dataset is obtained using following formula (Kreuter et al., 2001):

$$ESV = \sum(A_k \times UV_k) \qquad (4.1)$$

where ESV is the total value of watershed, A_k (ha) is the area and UV_k is the unit value ($/ha/year) for land use category k.

We compared global, regional and local datasets in two settings. First, the two global (global 1 and global 2) and a regional datasets are compared for valuation of all major biomes in the Grand River watershed where each biome is used as the proxy measure of value of all ecosystem services. Second, a global (Global 2), regional and local datasets are compared for the value of three ecosystem services from the biomes in the watershed.

In addition, the impact of the land use data of different spatial resolution (fine and coarse) is also assessed in terms of change in the value of ecosystem services.

Table 4.2: Area of land use categories in the Grand River watershed (GRCA, 2015). (The categories shown in *italic* are not valuated for ecosystem services. The numbers in bold are added up for total area and percentage of the main land use category).

Categories	Subcategories	Area (hectares)	Area (%)
Agriculture		**446,162**	**66**
	Row crops	133,082	19
	Small grains	79,662	12
	Forage	127,389	19
	Bare agriculture fields	106,029	16
Pasture/sparse forest		**55,660**	**8**
Forest		**72305**	**11**
	Dense forest (Deciduous)	35,722	5
	Dense forest (Conifer)	11,731	2
	Dense forest (Mixed)	19,497	3
	Plantation (Mature)	5,305	1
Wetlands		**64,278**	**9.5**
Open Water		**8,475**	**1**
Urban		**29,442**	**4**
Extraction	*(roads/beach/bedrock)*	***3256***	***0.5***
Total		**679,820**	**100**

4.3.2.1 Comparison of regional and global datasets for biomes

Kennedy (2014) suggests that if studied and policy sites share few characteristics, the benefit transfer will produce better results. Therefore, we developed a regional dataset collecting values from neighboring and regional valuated sites.

In this comparison, we assigned the unit values to all major biomes in the Grand River watershed from three datasets: global1, global 2 and regional dataset. A single unit value is assigned to each biome by aggregating the values of all ecosystem services it produces. To match the land use categories, proxy values for some of the land use categories (e.g., pasture) are used from similar or representative biomes in the respective dataset.

4.3.2.2 Comparison of local, regional and global datasets for ecosystem services

The global 1 dataset does not provide the unit values for the services for which primary estimates are available in the Grand River watershed. For example, carbon sequestration unit value is inferred from the primary data (local dataset) for the Grand River watershed, but the global 1 dataset leaves it out. Therefore, we excluded the global 1 dataset from this analysis and used only the global 2 as representative of global datasets. We compared the local, regional and global values to determine the accuracy of regional and global datasets with respect to local data or primary estimates. The unit values of only three ecosystem services are available for different land use categories in the Grand River watershed and these values are developed using cost-based approaches. Accordingly, local, regional and global datasets are compared for the set of three ecosystem services.

We use global (ESVD 2012), regional (mentioned above), and local (primary) datasets for unit values of ecosystem services generated from terrestrial biomes. The TEEB database, which is mainly based on ESVD 2012, is better for valuation of a limited number of ecosystem services because it contains standardized unit values for ecosystem services and sub services from different regional and global studies. It includes 1,310 unit values for different ecosystem services and these values are standardized to 2007 International dollars per hectare per year. This database is based on local studies across the World. De Groot — the lead author of the TEEB database — wrote a paper later in 2012, which is being widely used for global estimates (Table 4.1).

4.3.2.3 Impact of low and high spatial resolution of land use data

We assessed the impact of land use data with low and high spatial resolution on the value of ecosystem services in the Grand River watershed. Tailoring of data becomes inevitable in some situations depending on availability of unit values and resolution of land use data. Firstly, if the unit values are available only for major land use categories and not for sub-categories, then the values of major land uses serve as proxy for their sub-categories. Secondly, even the unit values are available for a variety of land use categories but land use data is coarse and contain only major land use categories, then the unit values are averaged for major land use category. Therefore, we investigated the impact of different resolutions of land use data on the value of ecosystem services by aggregating the subcategories under the major land use category and averaging the unit values to value it.

4.3.3 Coefficient of sensitivity

The coefficient of sensitivity (CS) is calculated for each land use category to show the percentage change in the total value with respect to percent change in the unit value (Kreuter et al., 2001; Wang et al., 2014; Zhang et al., 2015) and is calculated as:

$$CS = \left| \frac{{}^{ESV_j - ESV_i}\!/_{ESV_i}}{{}^{UV_{jk} - UV_{ik}}\!/_{UV_{ik}}} \right| \qquad (4.2)$$

where ESV_j and ESV_i are the adjusted and initial total values and UV_{jk} and UV_{ik} are the unit values for 'i' and 'j' datasets, respectively, and 'k' signifies the land use category.

The CS shows the proportional change in the total value with relative to the proportional change in the unit value of a land use category. The greater the proportional change in the total value, the more critical becomes the unit value. Therefore, CS predicts the veracity of the unit values. With higher CS, the use of an accurate unit value becomes more critical (Kreuter et al., 2001). However, Aschonitis et al. (2016) suggested that this approach can only be used for ranking the importance of a land use category in the total value of ecosystem services due to change in its unit value. They argued that CS values always remain between 0 and 1. Further, Aschonitis et al. (2016) proved that the use of this approach in assessing the robustness and sensitivity of unit values of ecosystem services is erroneous and must be abandoned (Aschonitis et al., 2016).

The change in ecosystem service values for land use data of different resolutions is calculated as (Song and Deng, 2017):

$$C_i = \frac{E_{end} - E_{start}}{E_{start}} \times 100 \qquad (4.3)$$

where C_i is the change in the total value of an ecosystem, E_{end} and E_{start} are the final and initial values, respectively.

4.4 Unit-value datasets

4.4.1 Global datasets

4.4.1.1 Global 1

The Costanza et al. (1997) dataset on unit values of ecosystem services is based on the value transfer method and includes the value of 16 biomes for 17 ecosystem services (Costanza et al., 2014). To obtain unit values, more than 100 studies are synthesized which are based on a wide variety of methods with underlying assumptions. The dataset also includes original calculations for a few ecosystem services. However, It is mentioned in the limitations that the dataset leaves out some of the ecosystem services provided by certain biomes, underestimates some major biomes (e.g., cropland) due to lack of information, and that willingness to pay may not reflect social fairness due to misinformation provided to individuals about ecosystem services (Costanza et al., 1997).

4.4.1.2 Global 2

We deduced the unit value of biomes and ecosystem services from the ecosystem services valuation database (ESVD) (de Groot et al., 2012) and those provided by Costanza (2014) based on ESVD. These values are referred as Global 2. This database (ESVD) incorporates the value of 10 biomes considering 22 ecosystem services for each biome based on local case studies across the world. The 22 ecosystem services are further divided into 90 sub services. In total, the database contains more than 1350 data points from over 300 case studies and 665 standardized values (value per hectare in 2007 international dollars) (de Groot et al., 2012). The cropland biome is not assigned a unit value for any of its services in the database. Majority of the values in ESVD are taken from the grey literature consisting of reports by experts with background in ecological economics (Schmidt et al., 2016). ESVD includes more number of studies (around 300 in total) than Costanza (1997) which had less than 100 studies (Costanza et al., 2014). Both databases are being used interchangeably for valuation of ecosystem services in different parts of the world (Table 4.1).

In ESVD, the climate mitigation is provided through carbon sequestration. Therefore, we used climate mitigation value as carbon sequestration. However, this database elaborates that the values of carbon sequestration has recently been recognized and not fully perceived, therefore these results can lead to undervaluation in important decision-making processes (de Groot et al., 2012). We used mean values from the global datasets for our analysis.

4.4.2 Regional dataset

Several regional studies were reviewed to establish compendia of values for ecosystem services/biomes analogous to the global datasets (e.g., global 1 and global 2). To build this dataset, we sifted through Environmental Reference Valuation Inventory (EVRI) and used other search engines (e.g., Google scholar and Web of Science) to find relevant information from regional valuation studies. Environment Canada developed EVRI (http://www.evri.ca) in 1990 to help analysts using benefit transfer method for assessing the value of environmental services. This inventory allows the users to scan and select the relevant regional studies (McComb et al., 2006).

Regional dataset is based on the unit values derived in the neighboring watersheds or regions. For this dataset, only those studies are selected that use totally or partially local data to generate unit values. Among these studies, a study focused on southern Ontario (Troy and Bagstad, 2010) is based on value transfer method; therefore, we exercised due care and picked out only regional unit values from this study.

We used the mean unit values (±SD) from the four regional studies (southern Ontario, Lake Simcoe, Ontario's wealth (green belt) and Peace River), which is a general practice in the valuation literature based on the value transfer method. Of these four studies, only Peace River area lies in the boreal region. This study is selected to make the unit values compatible with global data because global dataset (Global 2) aggregates the values for temperate and boreal forests. Other three locations are neighboring to Grand River watershed and lie in the temperate zone. The four regional studies, used for the value transfer, valuate a variety of land uses for different ecosystem services. The selected values of the ecosystem services are the average of the unit values given in the studies for these services.

4.4.3 Local dataset

There are two studies in the literature that focused on valuation of ecosystem services within the Grand River watershed (e.g., Aziz et al., 2017; Belcher et al., 2001). These studies are the first-hand monetary appraisal of ecosystem services in the watershed and have valued three ecosystem services, in total, based on land use categories (Schmidt et al., 2016). The available data from the literature are used to develop unit values for the primary valuation. The unit values derived applying cost based approaches (e.g., replacement cost and avoided cost methods) are only used for primary valuation. Both of the local studies used cost-based approach to value the ecosystem services and are therefore consistent.

The cost-based approach is appropriate for measuring a single or a limited number of ecosystem services when technological solutions cannot not generate all of the services that are provided by a given ecosystem (Notaro and Paletto, 2012). The basic assumption for the replacement cost method is that the proxy cost must not be greater than the benefit obtained from the ecosystem service. Otherwise, the replacement cost method may yield high values that could misrepresent the willingness to pay or the willingness to accept. Moreover, the replacement cost method ignores any other benefit the replacement substitute may be generating (e.g., employment opportunities, energy production, etc.).

The replacement cost method can be used to valuate non-market values if a perfect substitute is present with similar benefit provision as the original system. However, it is impossible to value cultural services using replacement cost method (Ledoux and Turner, 2002). Application of replacement cost requires three conditions to be met: perfect substitute means that human engineered system must provide an equivalent function in quality and quantity to the ecosystem service; least cost alternative means cheapest human engineered system to replace ecosystem service; and individuals must be willing to incur the cost in case of ecosystem service loss. In our opinion, the current market prices, which are used in replacement cost method, describe the consumers' willingness to pay. However, the review of replacement cost studies shows that these three conditions are rarely achieved.

We used local unit values which are derived using cost-based approaches, and local data of ecosystems and market prices. These local unit values of ecosystem services serve as baseline dataset because these values depend on local ecosystem and market data and are, therefore, locally relevant.

4.5 Results

4.5.1 Unit values of biomes from global datasets

The unit values of major biomes in the Grand River watershed are taken from the global datasets (Table 4.3). We obtained some of these unit values from Costanza et al. (2014) because it provides comprehensive, standardized and mean values for all biomes (such as cropland) for both datasets. Both global (1 and 2) datasets do not assign a unit value to pasture category; therefore, we used rangeland/grassland as a proxy land use for pasture. Similarly, the value of swamps/floodplains is used as proxy for inland wetlands and river/lakes for open water. The bare agricultural lands are given the same value as of cropland.

The values of different biomes from these datasets show noteworthy differences (Figure 4.1). The minimum difference is in the values of inland wetland and open water, and maximum difference is in the values of cropland.

Table 4.3: Unit values of terrestrial biomes based on global (Global and Global 2) and regional datasets (see sections 4.4.1 and 4.4.2 for details). All values are converted to CAD 2017.

Biome	**Unit values (2017 CAD/ha/year)**		
	Global 1	Global 2	Regional
Forest	665±130	4,725±1770	4,790±2315
Pasture	510	4,500±4230	1,725±1260
Wetlands	43,030±31,800	40,260±25,920	13,760±7500
Open water	18,675±15,050	6,690±6,175	2,075±2620
Cropland	200	8,730*	845±450
Bare agricultural lands	200	8,730*	2,050
Urban	-	10,450*	1,015

*values are taken from Costanza et al. (2014). In the global datasets, the bare agricultural land is assigned the same value as that of the cropland.

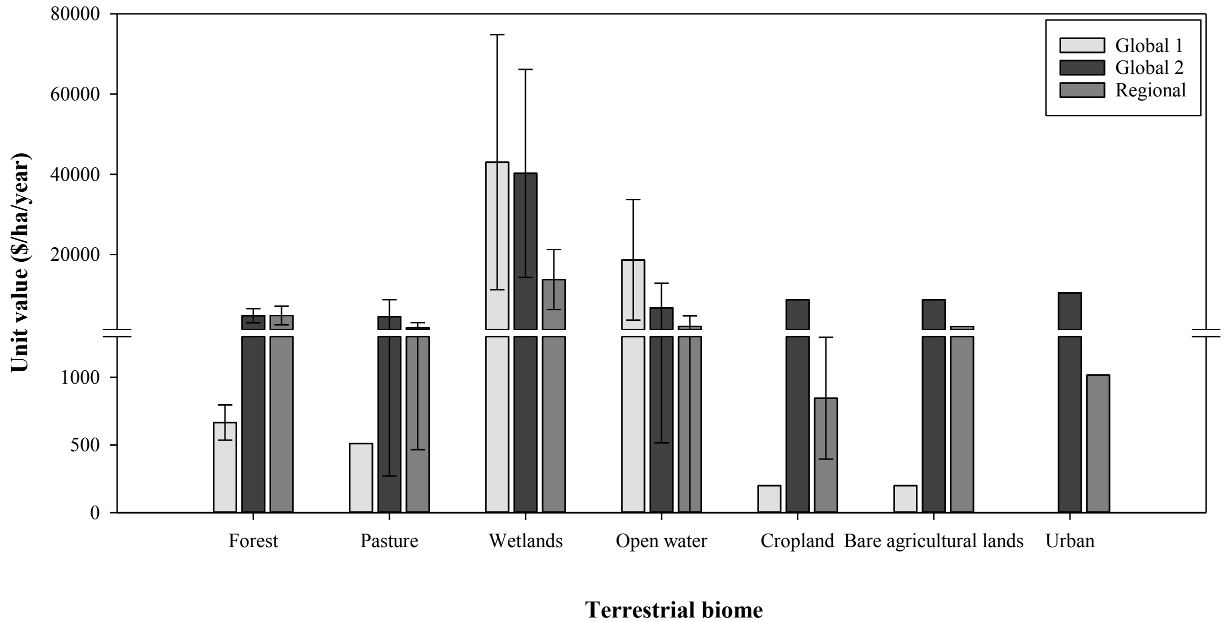

Figure 4.1: Range of unit values of terrestrial biomes from global and regional datasets. Global 1=value based on Costanza et al. (1997); Global 2=value based on ESVD (2012); Regional=value based on regional studies. Error bars show variation in the value of biomes. The values without error bars are based on single data point in the selected literature. All values are in CAD 2017.

4.5.2 Unit values of biomes from regional dataset

The unit values of different biomes in the Grand River watershed are deduced from the regional studies (Table 4.3). Four studies are selected from the regional database for this analysis. Three studies that focused on temperate regions, similar to Grand River watershed, are Greenbelt (Wilson, 2008b), southern Ontario (Troy and Bagstad, 2010) and Lake Simcoe (Wilson, 2008a) studies. The only boreal study was Peace River watershed (Wilson, 2014). These studies largely used the local or regional ecosystems and market data to value ecosystem services. Therefore, these estimates can be considered as reflection of local/regional ecological and economic conditions. These studies span over 6 years and values are scattered. Similar to the global dataset, there is a significant difference in the regional dataset in high and low unit values of a biome. However, this difference is of lower order of magnitude compared to the difference in the mean unit values from two global datasets.

4.5.3 Local, regional and global unit values of biomes for three ecosystem services

The primary unit values of only three ecosystem services were available for the Grand River watershed: carbon sequestration, nutrient cycling and water filtration. Therefore, for comparison of global, regional and local values, we selected the unit values of these three ecosystem services. The global unit values of ecosystem services are based on ESVD (2012) and only those studies are selected which yield the estimation in US or Canadian dollars. Further, we adjusted these values for consumer price index (CPI) and standardized to CAD 2017 using inflation calculators. Regional values are taken from a self-established dataset (see section 4.4.2) and local values based on replacement cost method are the best estimates in the Grand River watershed (described in section 4.4.3).

The regional studies usually assign a mean value to ecosystem services and did not include the standard deviation. However, we calculated standard deviation based on mean values from different studies.

These datasets did not assign unit values to certain biomes (e.g., agriculture) for these three ecosystem services due to either lack of data or those biomes do not generate the services (Figure 4.3). There is large variation in the unit values taken from global dataset compared to regional and local datasets. Some of the ecosystem services (e.g., carbon sequestration) and their importance was not recognized at the time of ESVD development; therefore those services are assigned lower unit values (de Groot et al., 2012).

4.5.4 Total value of watershed biomes based on global and regional dataset

The total value of the watershed for its ecosystem services based on biomes' unit values is obtained using the global and regional datasets (Figure 4.2). The value of the watershed is $3.30 billion/year and $7.85

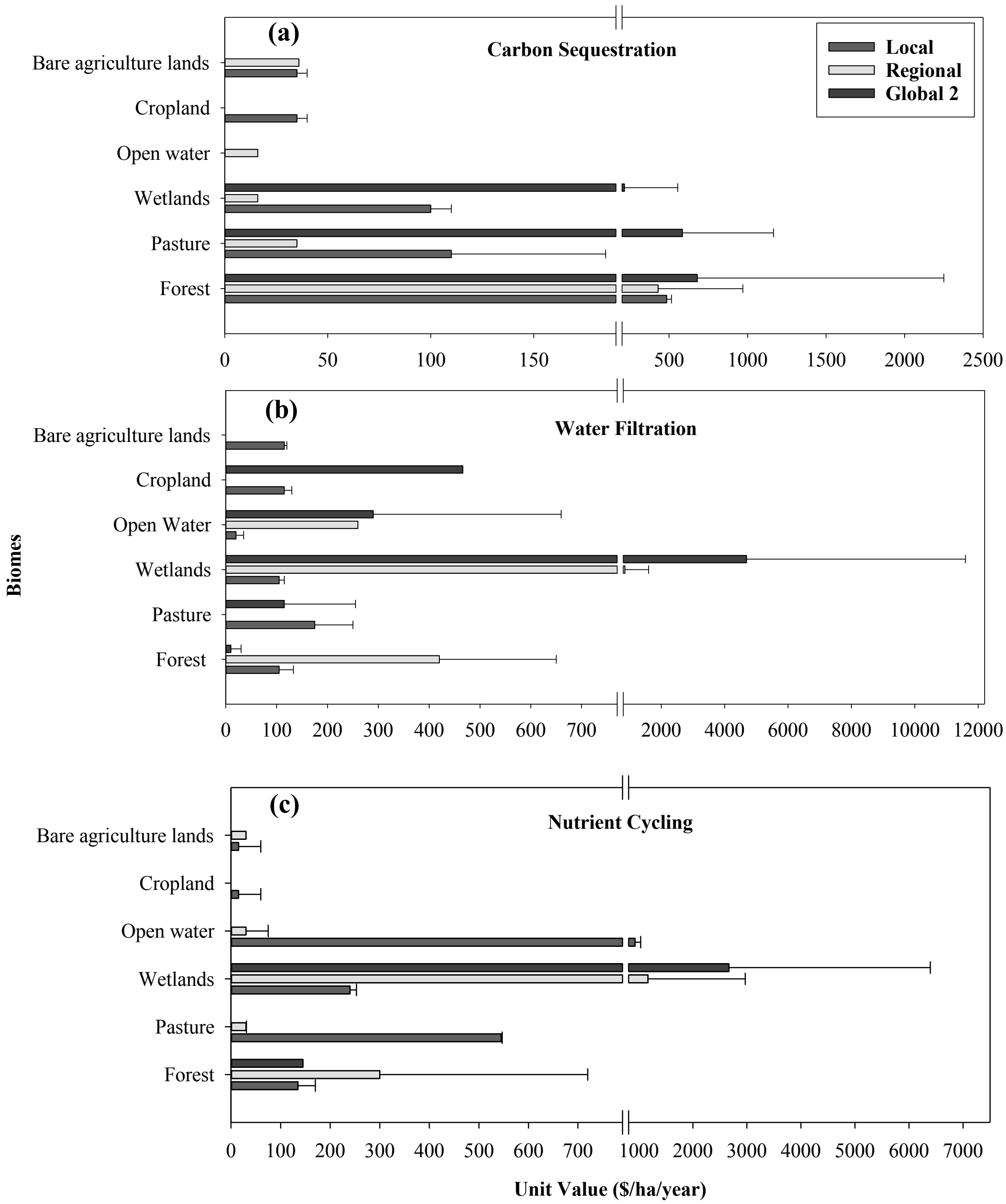

Figure 4.2: Unit values of a) carbon sequestration, b) water filtration and c) nutrient cycling, based on local, regional and global datasets. Error bars show variation in the unit values of ecosystem services for each biome. The values without error bars are based on single data point in the selected literature. All values are in CAD 2017.

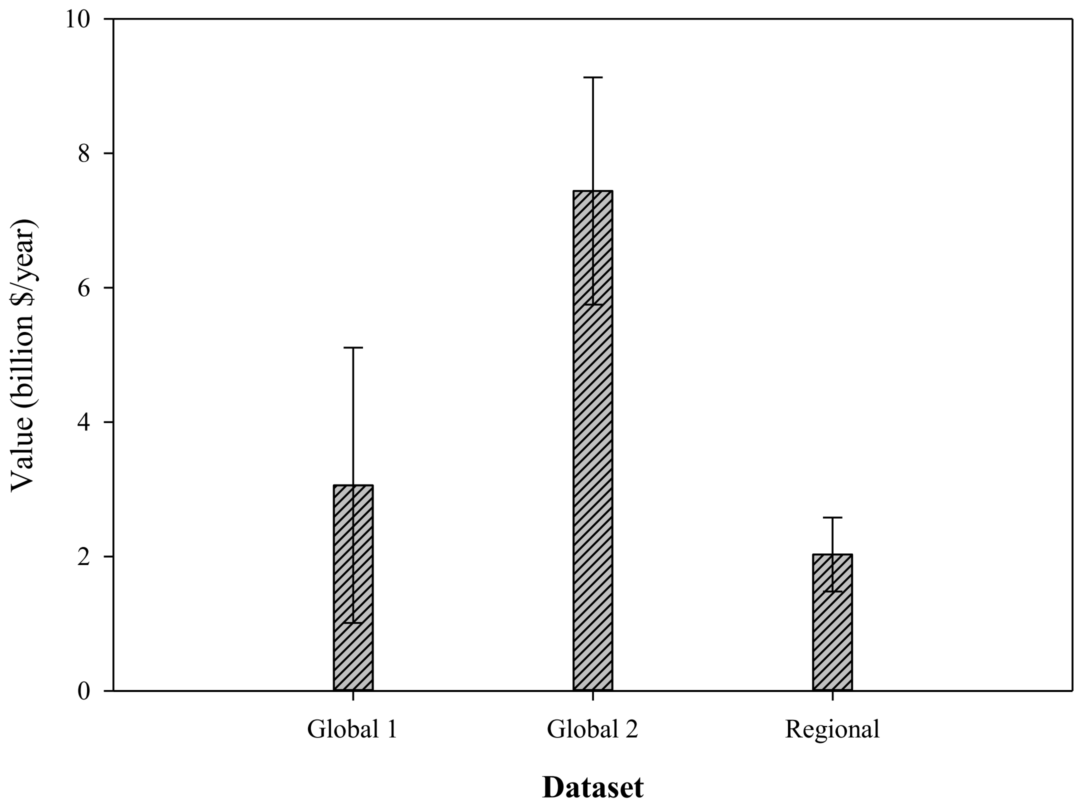

Figure 4.3: Total value of Grand River watershed for its ecosystem services based on global (Costanza, 1997; ESVD 2012) and regional datasets. Global 1=value based on Costanza et al. (1997); Global 2=value based on ESVD (2012); Regional=value based on regional studies . Error bars show variation in the values of the respective dataset. All values are in CAD 2017.

billion per year based on the global 1 and global 2 datasets, respectively. However, a value of $1.60 billion/year is obtained using unit values of biomes from the regional dataset.

4.5.5 Total value of the watershed for three ecosystem services based on local, regional and global datasets

The Grand River watershed is valuated for three of its ecosystem services based on local, regional and global datasets of unit values (Figure 4.4). The values of these ecosystem services based on local and regional datasets are in close agreement. However, the global datasets yielded higher values for each of the selected service, and resulting in 3.6 and 3.3 times a higher total value for the watershed compared to the values based on local and regional datasets, respectively.

4.5.6 Value of ecosystems for different land use resolutions

Generally valuation studies valuate the major land use categories based on available data on land use and unit values. The global databases of unit values also focus the major terrestrial land use categories. However, our primary study (Aziz et al., in prep.) on the Grand River watershed not only put forth a methodology for comprehensive valuation of non-consumptive ecosystem services but also developed the local unit values for all major land use categories and subcategories for three ecosystem services (water filtration, carbon sequestration, nutrient cycling). In that study, land use data of forest and agriculture categories was subdivided into three sub-categories each. The forest was subdivided into deciduous, coniferous and mixed forest whereas agriculture into row crops, small grains and forage. Further, we developed local unit values for each category which are used in this analysis as well. The unit values (in CAD 2017) for three ecosystem services for deciduous, coniferous and mixed forest are $1700±185, $855±105 and $1200±140 /ha/year, respectively and for row crops, small grains and forage are $230±155, $150 ±105 and $115±125 /ha/year, respectively.

We aggregated the area of subcategories into major land use categories to investigate the impact of low resolution land use data on the value of ecosystem services. Further, we averaged unit values of ecosystem services for subcategories of forest and agricultural land use to apply on the aggregated land use area. The averaged unit values for forest and agricultural land use are $1250±85 and $165±75 per hectare per year, respectively. Applying these unit values on the land use area (Table 4.1), the total value of forest and agricultural land use for high and low resolution in the watershed is calculated (Table 4.4). Furthermore, percentage difference was calculated between the two values for high and low resolution data, using Equation 3. The unit values for all major land use categories are developed for three ecosystem services (water filtration, carbon sequestration, nutrient cycling) in the Grand River watershed.

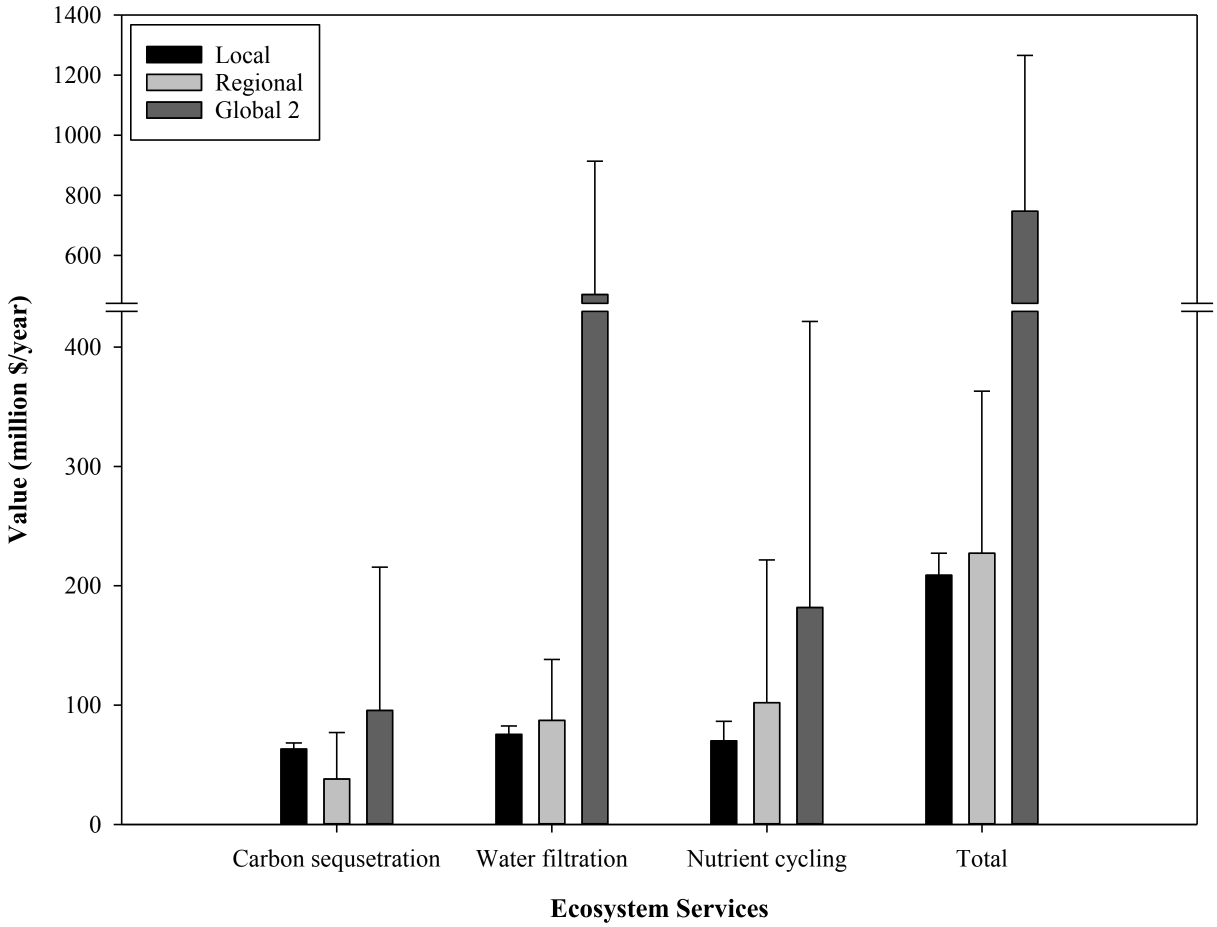

Figure 4. 4: Total value of three ecosystem services based on different datasets (global, regional, local) in the Grand River watershed. Global 2= based on ESVD (2012); Regional= values based on regional studies; Local= values based on studies conducted within the watershed. Error bars show variations in the values. All values are in CAD 2017.

Table 4. 4: Values of forest and agriculture categories based on low and high resolution land use data. Percent change is calculated using equation 4.3 (see section 4.3.3). The P values are taken from t-Test (given in the supplementary material).

Land use data	**Value (million $/year) of land use category**		
	Forest	Agriculture	Total
High resolution	94±7	57±27	151±28
Low resolution	84±6	56±26	140±27
Percent change, C_i (%)	**12**	**2**	**8**
P (T<=t) two-tail*	**1.07E-06**	**0.741**	**0.000533**

*** For P ≤ 0.05, there is a significant difference between the means. The details of t-Tests are provided in the supplementary material (supplementary tables AC 4-6).**

4.6 Discussion

4.6.1 Coefficient of sensitivity (CS)

The CS, calculated from the unit values of different datasets (Table 4.5), shows that the local and regional unit values are close to each other. Therefore, unit values of all land use categories are important in determining the total value of the watershed. Among all other datasets, the most important factor is the unit value for wetlands, which is the highest and, therefore, greatly influences the total value of the watershed. However, unit values of forests and wetlands are equally important in both the regional and global 2 datasets.

4.6.2 Differences in values across datasets

The main objective of this exercise was not to valuate the watershed but to compare different unit-value datasets to assess their reliability in comparison with regional and local datasets. Firstly, we appraised two global datasets (global 1 and global 2) with respect to a regional dataset. Further, the accuracy of the global and regional datasets is corroborated by comparing them with a local dataset using the unit values of three ecosystem services in the Grand River watershed.

The comparison of unit values of different terrestrial biomes from three datasets (Figure 4.1) showed that the global 1 dataset assigns lower unit values to most of the biomes except open water and wetlands which are attributed significantly high values relative to global 2 and regional datasets. The reason for lower unit values for most of the biomes in the global 1 dataset is the fact that the dataset leaves out many ecosystem services due to lack of information on those biomes. For example, temperate/boreal forests are assigned no value for water regulation, erosion control, nutrient cycling and habitat/refugia, but indeed these services are provided by temperate/boreal forests.

There are numerous reasons for variation in unit values between global 1 and global 2 datasets such as availability of new estimates, changes in functionality of ecosystems, and changes in human or built capital (Costanza et al., 2014). Costanza et al. (2014) re-estimated the value of global ecosystem services based on global 2 (ESVD (2012)) unit values, which yielded ~6 times higher value than estimates based on the globa1 dataset (both converted to 2007 international dollars) for same land use area of terrestrial biomes. Nevertheless, Costanza et al. (2014) considered global 2 dataset a better estimation/approximation because it was based on more complete and comprehensive set of ecosystem services. The values of wetlands (swamps/floodplains) and open water (lakes/river) showed minimum difference and it is explained by the reason that these ecosystems were well-studied in year 1997 when

Table 4.5: Coefficient of sensitivity (CS) for different land use categories (biomes) based on unit values from different datasets calculated using equation 4.2 (see section 4.3.3 for details).

Land use category	**Coefficient of sensitivity (CS) between different datasets**			
	Local and Regional	Regional and Global 1	Regional and Global 2	Global 1 and Global 2
Forest	0.25	0.02	0.37	0.02
Pasture/sparse forest	0.22	0.00	0.02	0.01
Wetlands	0.14	0.93	0.58	0.90
Open water	0.04	0.04	0.01	0.05
Cropland	0.27	0.03	-	0.02
Bare agriculture lands	0.08	0.01	0.03	0.01
Urban	-	-	-	0.10

global 1 dataset was established (Costanza et al., 2014). Despite all this, Costanza (1997) estimates have been criticized for overestimation of unit values for wetlands and underestimation for croplands (Wang et al., 2014).

For same biomes, the unit values in the global 2 dataset are generally higher than those in the regional dataset; however, both datasets assign approximately the same unit value to forests. Unit values of some biomes (e.g., pasture) are based on single data point (taken from a single study) and, therefore, there is no deviation in those values. The high variation in the unit values of biomes for individual and all ecosystem services is due to heterogeneity in the valuation studies. These unit values are taken from a number of studies conducted using a wide range of environmental resource valuation literature. de Groot et al. (2012) stated five reasons for large variation in unit values of global 2 (ESVD) dataset: a broader range of valuation studies from around the world, a variety of valuation methods, inclusion of a variety of subservices, possibility of double counting, and specificity of unit values with respect to space and time (de Groot et al., 2012). Yet, the range of unit values in regional dataset is significantly narrower than global 1 and global 2 datasets.

The total value of the watershed based on these three datasets (global 1, global 2, and regional) show that the global 2 dataset yields a highest estimate and the regional dataset a lowest estimate. The value of the watershed based on the global 2 dataset is 3.7 times higher than the value based on the regional and is 2.4 times higher than the value based on the global 1 dataset. However, deviation in the mean value of the watershed is largest for the global 1 dataset and smallest for the regional dataset.

The local data for three ecosystem services was available, which is used to validate the authenticity of the global and regional datasets. The global 1 dataset does not include carbon sequestration or its equivalent service; therefore, we used the global 2 dataset for this analysis. There is no clear trend in the unit values of three ecosystem services for terrestrial biomes between three datasets. However, deviation in unit values in the local dataset is much smaller compared to the regional and global 2 datasets (Figure 4.3). The global 2 dataset does not valuate some of the biomes (such as agricultural) but assigns higher unit values to others. Therefore, total value of the watershed for three ecosystem services based on the global 2 dataset is notably higher than the local (3.6 times) and regional datasets (3.3 times). On the other hand, total value of the watershed obtained using the local dataset is much closer to the value based on the regional dataset. This closeness in values may be due to the similarity of the biophysical and socio-economic characteristics between the regions of local and regional datasets (Feuillette et al., 2016). Further, locally relevant unit values resulted in considerably smaller variability/variation in the total value of watershed compared to the other datasets. The coefficients of variation in the total values based on the local, regional and global 2 datasets are 0.09, 0.60 and 0.69, respectively. These numbers reflect that the deviation in the mean values can be significantly narrowed down while moving from the global to the regional or local estimates.

The analysis of different land use resolution showed that aggregating the values of subcategories into a major category decreased the total value and its variation for a biome or a land use category. The forest land cover has greater unit value for three ecosystem services than agriculture but much less percent cover (11%) than agriculture (66%). However, the forest cover showed greater change in its value (12%) than agriculture (8%) due to higher unit value (Table 4.4). The results demonstrate that the land use data of fine resolution resulted in higher value of ecosystem services than the coarse resolution data, but the coarse resolution can decrease the range of values. Konarska et al. (2002) showed the similar results with more than 200% increase in the value of ecosystem services based on the fine resolution data. The reason for this huge increase was identification of different and more valuable ecosystem (e.g., wetland) in that study. Our results agree with the findings of Konarska et al. (2002) that the fine resolution data increases the value of ecosystem services, however this increase may not be that large. In our data, there is no change in the spatial extent of the major land use category; therefore, the only factor which influenced the value was aggregation of subcategories into the major category.

The global 1 dataset of ecosystem services was primarily developed and used for an awareness-raising exercise, which served its purpose successfully. These global estimates showed the importance of ecosystem services to human well-being in relevance to the other contributors (Costanza et al., 2014). However, numerous ecosystem services provided by different biomes are not valuated in this dataset due to lack of information (Costanza et al., 1997). These missing values resulted in underestimation of the biomes. The value estimate of three ecosystem services based on global 2 dataset (Costanza et al., 2014; de Groot et al., 2012) is markedly higher than values based on local and regional data. It can be concluded that the global estimates (global 1 and global 2) are useful in highlighting ecosystem services but lack a specific decision-making context (Costanza et al., 2014).

Simpson (2016) argues that the value transfer method is unreliable and meaningless because it fails to capture relevant/local context. A hectare in one location with high value does not imply that the same hectare will have high value at another location. In contrast, Costanza et al. (2014) supported the benefit transfer method with the argument that it is similar to the approach used for Gross Domestic Product (GDP) accounting where price is multiplied with quantity for each sector of the economy. However, they emphasized the use of regional aggregates for appraisal of land use change scenarios, national aggregates for national income accounting, and global aggregates for awareness-raising purposes

Table 4.6: Local, regional and global (Global 2) unit values of terrestrial biomes for three ecosystem services.

Ecosystem Services	**Providing Land Covers**	**Value (2017 CAD/hectare/year)**		
		Local	Regional	Global
Carbon Sequestration	Forest	485±30[a,b]	430±540	680±1570
	Pasture	110±75	35	585±580
	Wetlands	100±10[a,b]	16	215±340
	Open water	-	16	-
	Cropland	35±5	-	-
	Bare agriculture lands	35±5	36	-
Water Purification	Forest	105±28[a,b]	420±230	10±20
	Pasture	175±75	-	115±140
	Wetlands	105±10[a,b]	850±750	4690±6910
	Open Water	20±15	260	290±370
	Cropland	115±5	-	466
	Bare agriculture lands	115±5	-	-
Nutrient cycling	Forest	135±35	300±420	145
	Pasture/sparse forest	545±2	30±1	-
	Wetlands	240±13	1170±1800	2665±3730
	Open water	930±105	30±45	-
	Cropland	15±45	-	-
	Bare agriculture lands	15±45	30	-

[a](the unit values presented in chapter 2 of the thesis); [b](Blecher et al., 2001)

(Costanza et al., 2014). In the same vein, our paper posits that the value transfer method can produce better results if unit values would be obtained from local studies. Doing so would help to capture the marginal values of ecosystem services in the regional context. When the values are transferred from the global to local scale, marginal values become highly erroneous because of the scarcity or abundance of a land use neglected in the two settings. For example, the unit value from a land use abundant in one place would be lower compared to another place where it is scarce (Simpson, 2016).

The estimation of ecosystem services based on the global datasets is a crude approximation due to loopholes in the data, absence of a stringent classification framework for ecosystem services, use of a variety of economic methods, and variability of spatial and temporal scales. The use of the global datasets is less reliable because the price shifts are unpredictable from place to place. Our results show that unit values from the regional dataset, which are partially based on the local dataset, yield a value much closer to the value based on the local dataset of ecosystem services as compared to the global dataset. Therefore, we concluded that the valuation based on the value transfer method is more meaningful if it uses the regional dataset rather than the global dataset.

Even though the total value of three ecosystem services based on the regional dataset is much closer to the local dataset, variation is higher in the total value based on the regional dataset. This variation can be narrowed down by using local data for unit values of ecosystem services (as explained in paper 1). Because the unit values depend on the time and location (EPA, 2015), they vary with change in these factors.

The global and regional datasets are based on a number of studies; therefore, the value transfer method was a realistic choice for comparison of these datasets. The benefit function transfer is not used because it needs adjustment of transferred function from a study to a policy site whereas each unit value from the global dataset is based a number of study sites which lack information required for function transfer.

The analysis of the land use data (or impact of land use refinement) shows that the coarser resolution data leads to a lower watershed value, whereas the finer resolution data results in a higher value. In addition, the resolution of high-valued ecosystems can significantly impact the total value of the watershed. For example, forest cover has a small area but a higher unit value; therefore, the percent difference in the value of ecosystem services for high and low resolutions of land use data is 12%. On the other hand, agricultural land use, which is 66% of the total area of the watershed but has a lower unit value compared to that of the forest, showed a difference of 2% in the total value of ecosystem services for low and high resolution data. Therefore, these findings are more specific to this study; changes in the area of high- and low- valued land use categories may lead to different results.

In this analysis, the cost-based approach is considered as a baseline method of valuation because it uses contemporaneous local data on market prices and ecosystems and also involves a market mechanism, that is, it uses market values as proxies for the valuation of the ecosystem service of interest. For comparing value transfer estimates based on the regional and global datasets, the value of the watershed for three of its ecosystem services obtained from the cost-based approach serves as the baseline estimate.

The global datasets are particularly focused on natural ecosystems and their services. Therefore, these datasets paid little attention to human-managed ecosystems (e.g., agriculture). However, human-managed ecosystems comprise a large area of the Earth's surface (Birkhofer et al., 2015). Similarly, the dominant land use in some watersheds is agriculture (i.e., the dominant land use in the Grand River watershed is agriculture which makes up 66% of the total area of the watershed). Thus, value estimates that use the global datasets tend to underestimate the dominant agricultural ecosystems and overestimate the natural ecosystems in the watersheds. In the regional and local datasets, agro-ecosystems are assigned higher values compared to global datasets; even then, the total value of the watershed based on the global datasets is higher due to a significant overestimation of natural land uses (Figure 4.3).

Sometimes, transferring data from global to local scales bring glaring discrepancies. For example, both the global 1 and global 2 datasets assigned a higher unit value to open water compared to the regional dataset (Figure 4.1). This higher value may be true at the global level but may not be that relevant for water-abundant regions like Canada. The consideration of such discrepancies in the analysis can help further improve ecosystem services estimates.

4.6.3 Variation of unit values of terrestrial biomes across regions/continents

We aggregated the standardized unit values (in 2007 international dollars) of different terrestrial biomes from ESVD dataset (global 2) for different regions (Figure 4.5), which showed significant variation from region to region. It supports our assertion of the need to develop the regional datasets for the accurate economic valuation of biomes/ecosystems located in different regions.

4.6.4 Limitations

The comparison of the regional and global datasets with the local dataset is based on a limited number of ecosystem services. Therefore, it needs further corroboration from a larger set of ecosystem services for better and reliable comparison between different datasets.

One key limitation of the value transfer method is that it is often oversimplified and this oversimplification is one reason which makes policy- and decision-makers indifferent of these estimates. For example, the value transfer method assumes that the supply of ecosystem services is constant in space

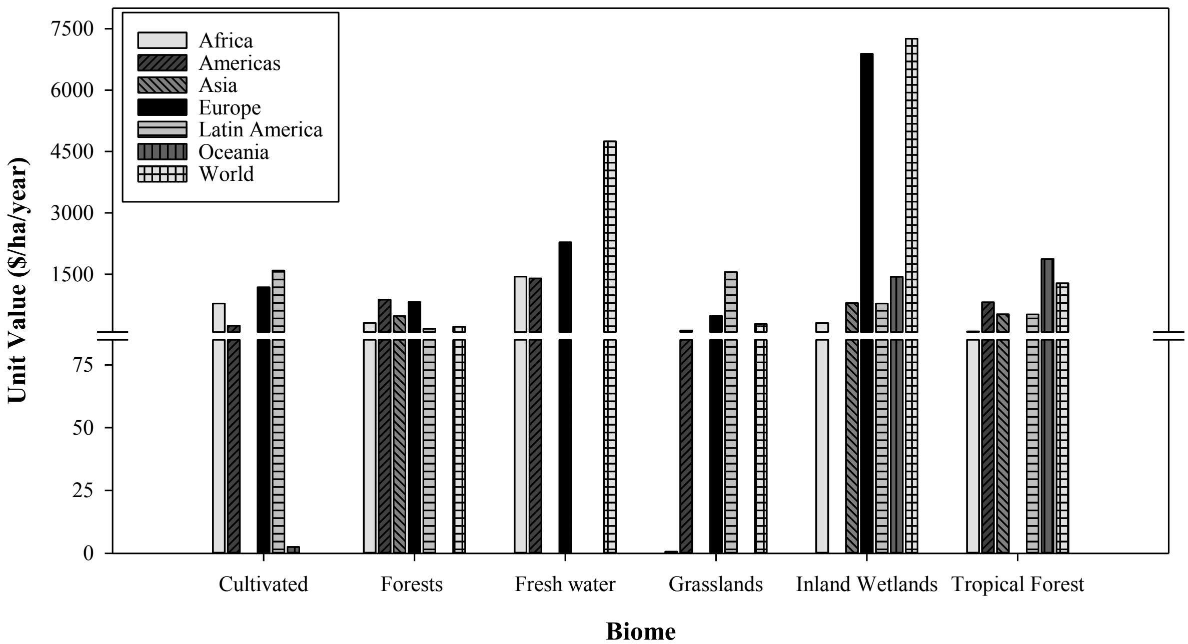

Figure 4.5: Unit values of terrestrial biomes for different regions based on ESVD (2012) database. "Forests" represent temperate and boreal forest biomes. All values are in CAD 2017.

and time between study and policy site (Whitham et al., 2015). These assumptions raise the question of validity of resulting value estimates. Thus, incorporation of variabilities in space and time can enhance the applicability of results of the value transfer method.

Furthermore, the absence of comprehensive global and regional datasets on unit values for all biomes in the Grand River watershed, and accessibility of limited local data on ecosystem services, led to an imperfect quantification of the regional and global datasets. Therefore, we recommend an analysis based on a complete set of ecosystem services to validate these preliminary results.

The two major errors are associated with value transfer approach: errors in the development of unit values at the study site and transfer errors. The transfer errors are space and time based errors (Navrud, 2004). Further, Navrud (2009) recommended the use of an average error bound of ±100% based on the validity tests of the value transfer.

4.7 Conclusions

The value estimates, based on unit values of biomes, from the global 1 dataset are closer to that of the regional dataset. Specifically, watershed values for its biomes based on the global 1 and global 2 datasets are 1.5 and 3.7 times higher than the regional dataset, respectively. However, unit values for most of the biomes in the global 1 dataset are underestimated because this dataset elides many ecosystem services provided by the biomes due to lack of information. Therefore, neither of the global datasets is reliable for an appraisal of ecosystem services at the regional scale.

The local dataset of unit values was available for only three non-market ecosystem services. These unit values are the primary/first-hand monetary estimates in the Grand River watershed. The unit values are based on locally-relevant data and have lowest variation/ standard deviation of the mean values. Therefore, the unit values of ecosystem services based on the local dataset were considered the most accurate and represented as the baseline dataset in comparison with other datasets. In this analysis, we found that the regional dataset is the second most accurate dataset because it yielded the closest value to the local estimate.

Our results showed that the global datasets led to very different outcomes for the estimation of ecosystem services compared to the regional and local datasets. This may be due to the fact that valuation is a space and time-based phenomenon and it changes with the change in location and time. Global values may be good for crude estimation of ecosystem services for awareness-raising but not for policy making. When the values are transferred from the global scale then marginal values become highly erroneous. The unit value from a land use abundant in an area would be lower compared to a land use which is scarce in another area (Simpson, 2016). Simpson (2016) argues that the value transfer is unreliable and meaningless due to its failure in capturing of relevant/local context. If a hectare in one location has a high value does not imply that the same hectare will have the same value at another location. The regional data will help to capture the marginal values of ecosystem services in the regional context. Similarly, the value transfer method can produce better results if unit values would be obtained from regional studies. Therefore, the value estimates based on regional ecosystems and market data can better inform policy- and decision- makers on monetary value of ecosystem services than global datasets.

The understanding of complicated natural processes, and measurement of indicators that can be conformed to economic analysis, underpin the value of ecosystem services. It can be concluded that the unit values (or value coefficients) derived at one time may not capture the value accurately at another much later point in time because of improvements in data, advancements in science and availability of up-to-date knowledge. The temporal difference between the regional and local dataset is minimal. Similarly, the total values of watershed based on the regional and local datasets are in close agreement.

Our results demonstrated that land use resolution can significantly influence the value of the ecosystems that have high variation in unit values for their sub-categories. Therefore, land use data of finer spatial resolution will help better estimate the value of ecosystem services. Despite plenty of information on the value of ecosystem services, we still lack the reliable information/methods.

This study offers a modest contribution towards understanding the role of different datasets in the economic valuation of ecosystem services and in making valuation science more meaningful. Clearly, further research based on a complete set of ecosystem services will be needed to validate the authenticity of regional dataset. However, we are able to conclude from this study that establishing and using regional datasets of unit values of ecosystem services will bring accuracy to the value estimates and facilitate their incorporation in the decision-making process at regional level.

In short, the following conclusions can be drawn from this study:

- Primary estimates are expensive and time consuming but are more reliable and narrow down the variation in the estimated total value of ecosystem services in the watershed.
- The regional dataset yielded estimates closer to the primary dataset and reduced the range of values for a watershed compared to the global dataset. Therefore, the use of regional datasets can increase the validity of the value transfer approach and can make it more meaningful at the regional and local levels.

- Global datasets may be used for preliminary appraisal for awareness-raising and assessing the changes in ecosystem services but not for policy- and decision-making at the regional and local scales.
- The fine resolution land use data resulted in higher and coarse land use data led to lower value estimates of ecosystem services.

Chapter 5

Moving well-being ahead: Value of potential and realized ecosystem services in southern Ontario, Canada

5.1 Summary

The full supply of services an ecosystem can generate are called potential ecosystem services; the fraction of the potential ecosystem services that is actually used by society are referred to as the realized ecosystem services. Because they are contributing to human well-being, the realized ecosystem services are of particular importance to people and, hence, to policy-makers. However, one of the key challenges faced by the economic valuation of ecosystem services is: How do we differentiate between the values of realized and potential ecosystem services? This project addresses this challenge by estimating the potential and realized values of ecosystem services in southern Ontario, which is the most densely populated region in Canada. We use a phenomenological model to determine the spatial distribution of a use index that varies between 0 (minimum use) and 1 (maximum use) for a bundle of six ecosystem services. We further derive unit values (in units of Canadian dollars per hectare per year) for the selected ecosystem services, based on the value transfer method. The estimated average potential value of the bundled ecosystem services is then \$19 billion per year for the southern Ontario region. To valuate the realized ecosystem services, the potential values are scaled by the corresponding use index values. The resulting average value of the realized ecosystem services is \$9.7 billion per year, that is, about 50% of the value of the potential ecosystem services. The distribution map of realized ecosystem services can help land use planners locate areas that could be targeted for investments in natural infrastructure.

5.2 Introduction

The presence of people is the precondition to accruing benefits from nature's ecosystem services (Costanza et al., 2014). Therefore, the value of used ecosystem services depends on their direct or indirect consumption by the people which is a purely anthropocentric utilitarian concept (François et al., 2005; Goldenberg et al., 2017; Jones et al., 2016). The supply of an ecosystem service is defined as the capacity or potential of an area to provide ecosystem services; demand is defined as the sum of ecosystem services consumed or used in an area (Burkhard et al., 2012). Usually, valuation studies value the production (supply) of ecosystem services and not the consumption (demand). Recently there is an increasing realization and wide agreement to capture the value of ecosystem services that are consumed in a specific area (Burkhard et al., 2012; Goldenberg et al., 2017; van Jaarsveld et al., 2005).

The total ecosystem service supply from an area is called potential ecosystem services and the whole or part of the potential ecosystem services consumed is defined as realized ecosystem services (Goldenberg et al., 2017). In economic valuation literature, most of the studies estimate the value of the supply side of ecosystem services and not of the consumed/realized services which is based on their demand in an area.

Although some studies conceptualize the idea of distinguishing potential and realized services (e.g., Fisher et al., 2008; Goldenberg et al., 2017; Syrbe and Walz, 2012), there is a paucity of studies (mapping and valuation) which use this concept practically.

There is no single agreed upon definition of ecosystem services. This raises confusion about implementing the outcomes of ecosystem service studies due to conflicting definitions and a variety of methods and approaches leading to contradictory results (Wainger and Mazzotta, 2011). Additionally, valuation studies of ecosystem services are void of information about the value of realized ecosystem services. However, information on the flow and use of ecosystem services would help decision makers and landscape planners in their decisions regarding land use planning and management. It would also be helpful to identify the provider areas upstream for the payment of ecosystem services used by the user living downstream (Fisher et al., 2008). Thus, the lack of clarity in defining realized ecosystem services hinders the integrated quantification (Jones et al., 2016) and hampers the implementation of outcomes of valuation studies.

The relationships between potential and realized ecosystem services differ based on the nature of ecosystem services. For example, carbon storage and sequestration services are used globally and therefore most of the potential supply of these services is realized. On the other hand, the water provisioning service by an ecosystem is directed downstream and it will not be realized unless the water is used by the people, businesses or agriculture situated downstream. Similarly, if there is less use of an ecosystem service then only a fraction of its potential service is realized (Mulligan and Clifford, 2015). Therefore, an area of high value for potential services may not be of high value in terms of realized ecosystem services.

Because realized ecosystem services are used by the people, their values have a real economic impact (Fei et al., 2018). Therefore, valuation of realized ecosystem services can strengthen the understanding of the geographical context and significance of different ecosystems. For example, a forest located in the remote wilderness provides much less realized ecosystem services compared to a forest situated in the vicinity of an urban area. Furthermore, the value of realized ecosystem services will help to protect the natural ecosystems in urban and peri-urban areas (Mulligan and Clifford, 2015).

The complete information on ecosystem services can help policy makers to meet the challenge of sustainability (Bennett and Chaplin-Kramer, 2016). The need for information on ecosystem services is emphasized in many policy documents (e.g., the sustainable development goals (SDGs) and Aichi Targets for biodiversity) for sustainable development. However, the recommended information for sustainability is biased towards the supply side of ecosystem services. On the other hand, there is much less information available on demand and use of ecosystem services. The demand and use for an ecosystem service varies with population, location and time, and is different for different ecosystem services (Geijzendorffer et al., 2017). Therefore, the information that captures the demanded and used ecosystem services can assist the decision makers towards achieving sustainable development goals. Furthermore, recognizing the demand can assist to understand the impact of ecosystem services on human well-being (Wei et al., 2017).

The mismatch of supply and demand of ecosystem services can have direct and indirect (often negative) impacts on human well-being. For example, the reduction in air purification in a city can impact the lives of its citizens directly, whereas global climate change resulting from reduction in carbon sequestration at one place can impact the human societies indirectly (Baró et al., 2015; Wei et al., 2017). The assessment of these mismatches between supply and demand of ecosystem services can be carried out by making a distinction between potential and realized ecosystem services, which can help enhance human well-being (Baró et al., 2015). Further, a distinction between potential and realized ecosystem services can help with land use planning, payment for ecosystem services, and their efficient management and use (Silvestri, S., Kershaw, 2010; Wei et al., 2017).

For economic valuation, it is vital to consider that an ecosystem service is a potential service until it is being used by the beneficiaries, at which point it becomes a realized service. Therefore, the realized ecosystem service is the function of a potential ecosystem service. Realized service, however, can be increased by benefitting more people through careful use and improved management without changing potential service, but cannot exceed the potential service in amount (Jones et al., 2016). Accordingly, the monetary value of realized ecosystem services heavily depends on the number of its beneficiaries and in this manner differs from the value of potential services.

However, capturing used ecosystem services (i.e. demand side) by carrying out spatially-explicit analysis is currently identified as a key challenge in the literature (Castro et al., 2014). We selected a region in southern Ontario to implement a methodology of differentiating the value of realized from the value of potential ecosystem services. There are several studies that valuate ecosystem services of this region or watersheds located within the region (e.g., Kennedy and Wilson, 2009; Troy and Bagstad, 2010; Wilson, 2008a, 2008b), but to our knowledge, no study has made a distinction between potential and realized ecosystem services. In this paper, we discern the realized ecosystem services from potential ecosystem services in southern Ontario's landscape. Further, we reflect the distinction of potential and realized ecosystem services in terms of their monetary values.

5.3 Materials and methods

5.3.1 Study Area

The appropriate scale to assess the potential and realized ecosystem services is landscape level (Castro et al., 2014). Within the landscape of southern Ontario, we selected those watersheds which are managed by conservation authorities and for which current land use data was available. Southern Ontario Land Resource Information System (SOLRIS) provides land use data updated to 2016 for ecoregions 6E and 7E. Several key initiatives in the province such as source water and natural spaces protection, and biodiversity conservation are based on SOLRIS data (MNR, 2008). This land use data completely covers most of the area managed by conservation authorities in southern Ontario. However, watersheds managed by Mississippi valley, Rideau valley, Quinte and Crowe Valley conservation authorities are partially covered by the land use data. Therefore, we selected only those areas which fall under the jurisdiction of conservation authorities and are covered by the SOLRIS data.

The selected area (Figure 5.1) is the most densely populated area in Canada and, therefore, the majority of its natural ecosystems are converted to other human uses such as urbanization and agriculture. In this area, major cities include Toronto, Kitchener-Waterloo, London, Kingston, Ottawa, Hamilton and Windsor, and major river systems include the Grand, Thames, Credit and Humber Rivers (Crins et al., 2009).

The SOLRIS data divides land use into 28 total categories (major and subcategories) for the selected region. For valuation purposes, we aggregated subcategories into six major land use categories (Figure 5.1): Forest includes treed cliff and talus, mixed, deciduous, coniferous, hedgerows and plantations; grassland includes open and treed alvar, tallgrass prarie, tallgrass savannahs and tall woodlands; wetlands include treed swamps, thicket swamps, fens, bogs and marshes; open water includes lakes, reservoirs and rivers; agriculture includes tilled and undifferentiated agricultural features; and extractions include transportation, built-up, pits, and peat soil. The total area of this region is 7,436,083 ha with agriculture as the dominant land use and it makes up 61% of the total area (Table 5.1).

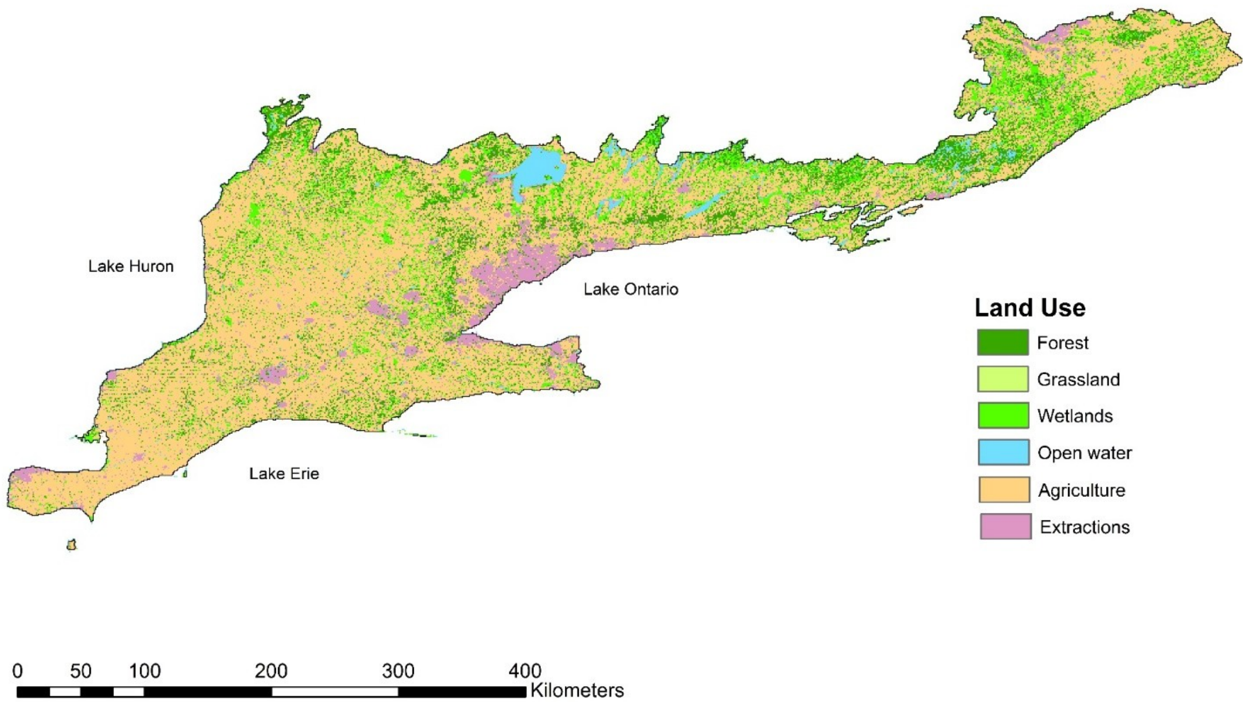

Figure 5.1: Land use in the study region of southern Ontario. The land use subcategories from the original land use data (MNR, 2008) are aggregated into six major land use categories.

Table 5.1: Area of land use categories in southern Ontario, Canada.

Land Use	Area (hectares)	Area (%)
Forest	1,021,638	14
Grassland	4,302	0.1
Wetlands	982,312	13
Open water	235,474	3
Agriculture	4,512,295	61
Extractions	680,062	8.9
Total	**7,436,083**	**100**

5.3.2 Methodology

We relied on/used an off-the-shelf tool to capture the value of supply and demand of ecosystem services in southern Ontario. We used Co$ting Nature, an ecosystem services mapping tool, to model the potential and realized ecosystem services in the selected region. This web-based tool uses pre-loaded global datasets to capture the spatial distribution of water, carbon, hazard mitigation, and nature-based tourism services. Further, it identifies/estimates the potential and realized ecosystem services and aggregates them into bundled services indices with values from 0 to 1 (Bagstad et al., 2013).

Co$ting Nature has biophysical models at the core for assessment of ecosystem services. It uses GIS databases and hydrological models to capture the complex hydrological processes, their ecosystem services, and the consumption of these ecosystem services at dam-, urban- and agricultural- sites (Silvestri, S., Kershaw, 2010). However, Co$ting Nature does not estimate the monetary value of ecosystem services and therefore cannot be used for direct valuation of these services (Bowles-Newark et al., 2014). The Co$ting Nature model takes the magnitude and geographic pattern of ecosystem services as potential ecosystem services and their use at local and global scale as realized ecosystem services (Mulligan, 2015).

The Co$ting Nature model was set up to run two tiles to capture the realized ecosystem services over the entire selected area. Finally, we performed the following steps to incorporate the results of realized ecosystem services obtained from the Co$ting Nature model into economic valuation:

1- We created a “big raster” (using QGIS) by merging realized ecosystem services rasters (maps/tiff files)
2- We used “dissolved” function to create the boundary/border for the selected area
3- Using the border, we extracted by mask the realized ecosystem services from the big raster
4- The land use layer was exported to a tiff file that has same cell size as the big map of realized services
5- We extracted by mask the land use layer with the border area
6- We rescaled the realized services raster values/indices (big map) from 0 to 1
7- We reclassified the rescaled realized services from 1 to 10
8- Next, we converted the reclassified and rescaled realized services raster to polygon
9- Finally, we used the zonal histogram tool to create a count of land use cell values in the realized services area polygon.

The map shows the areas from where people are using most of the selected six services in the region. To verify the model, we valuated the area for its potential for ecosystem services. We selected only those ecosystem services for valuation that are modeled by the Co$ting Nature model. To show the difference in the values of potential and used ecosystem services, the following ecosystem services are selected:

- Water provisioning/supply and quality
- Carbon sequestration and storage
- Flood regulation (hazard mitigation)
- Nature based tourism (recreational and aesthetic values)

We used the value transfer method and unit values to valuate the area for selected ecosystem services. The unit values are taken from the regional studies conducted in southern Ontario and used to estimate the total value of potential ecosystem services using equation (5.1) (Kreuter et al., 2001):

$$ESV_p = \sum(A_k \times UV_k) \qquad (5.1)$$

where ESV_p is the total value of potential ecosystem services, A_k is the area (ha), and UV_k is the unit value ($/ha/year) for land use category k.

The realized ecosystem services have different distribution across the region compared to potential ecosystem services. We averaged realized ecosystem services indices and named them RS1 to RS10. Further, we used these indices, unit values of ecosystem services, and land use areas to estimate value of realized ecosystem services by using equation (5.2).

$$ESV_r = \sum(RSI_i \times A_k \times UV_k) \qquad (5.2)$$

where ESV_r is the total value of realized ecosystem services, and RSI_i is the realized service index of an average value 'i'.

5.4 Results

5.4.1 Valuation of potential ecosystem services

We valuated the ecosystem services in the selected area using the value transfer method. The unit values are taken from the four studies conducted in the regions located in Ontario. These unit values are converted to CAD 2017 by adjusting for inflation (Table 5.2).

The unit values for open water and wetlands are higher than other land use categories (Figure 5.2) because these two land use categories generate a relatively higher magnitude of the selected services.

Using these unit values, the total value obtained for ecosystem services is \$19±0.8 billion/year.

5.4.2 Modeling of potential ecosystem services

Using Co$ting Nature, we modeled the potential ecosystem services in southern Ontario's landscape. The indices are rescaled between 0 and 1. The higher the index, the greater the potential of the area for ecosystem services (Figure 5.3).

5.4.3 Comparison of potential ES distribution with unit value distribution

The total unit values' distribution is similar to the potential ecosystem services' distribution in magnitude/frequency (Figure 5.3 & AD1). The higher unit value of an area implies its higher potential for ecosystem services and vice versa.

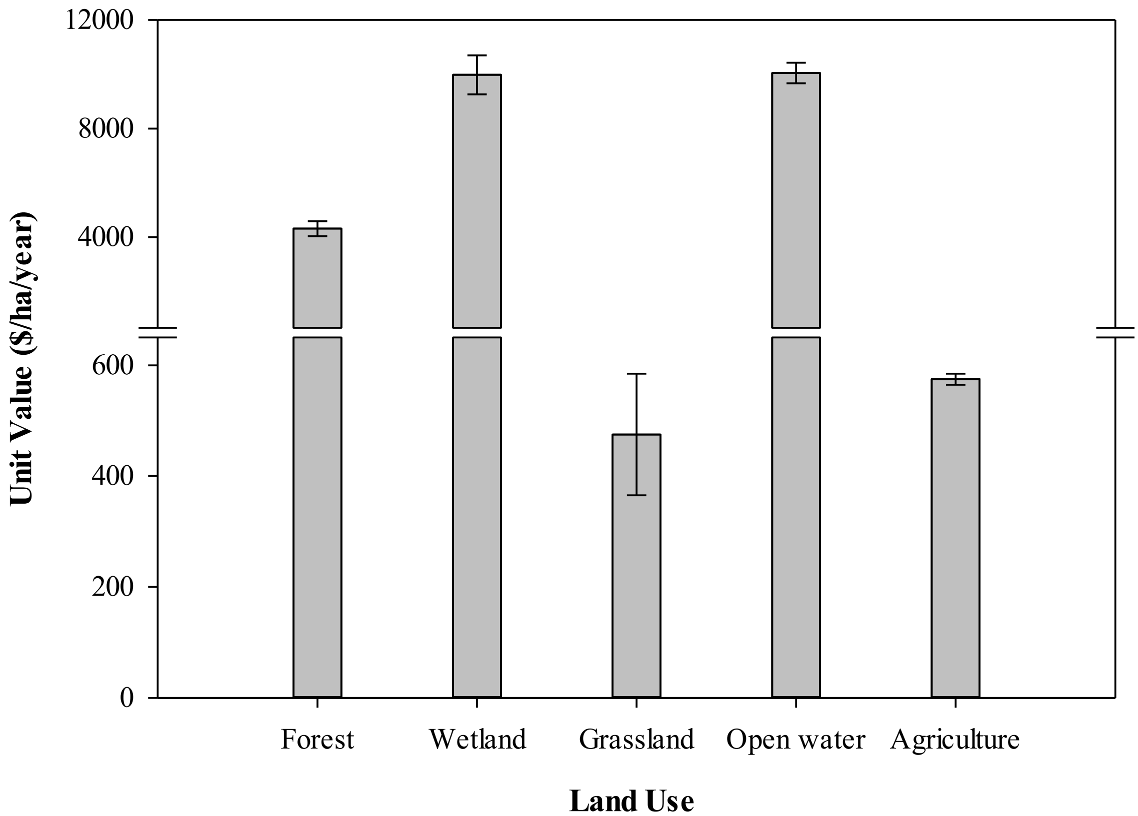

Figure 5.2: The total regional unit values ($/ha/year) assigned to land use categories for a bundle of six ecosystem services.

Table 5.2: The regional unit values of land use categories for a bundle of six ecosystem services.

	Unit Values of Ecosystem Services ($/ha/year)						
Land Use	Water supply	Water quality	Carbon sequestration	Carbon storage	Flood regulation	Recreation	**Total**
Forest	300±265[1,3,4]	595[2]	48[3]	1130[3]	1875[3]	360±70[2,3]	**4310±275**
Wetland	265±330[2,4]	3470±345[2,3]	16[3]	865±540[3]	4970[3]	410[3]	**9995±720**
Grassland	60[3]	105±107[2,3]	35[3]	260[3]	10[3]	5[3]	**475±110**
Open water	280±290[1,3,4]	3710[3]	15[3]	830[3]	4970[3]	240±250[2,3]	**10045±380**
Agriculture	-	-	-	410[3]	-	165±10[2,3]	**575±10**

([1]Kennedy and Wilson, 2009; [2]Troy and Bagstad, 2010; [3]Wilson, 2008a; [4]Wilson, 2008b)

Table 5.3: The averaged indices (from Figure 5.3) and total land use area (in hectares) falling under realized ecosystem services indices from RS1 to RS10.

	RS1	**RS2**	**RS3**	**RS4**	**RS5**	**RS6**	**RS7**	**RS8**	**RS9**	**RS10**
Averaged index (RSI)	0.14	0.23	0.32	0.41	0.50	0.59	0.68	0.77	0.86	0.96
Land Use area										
Forest	189	11,883	102,172	277,215	239,993	158,067	134,929	65,201	25,339	1,509
Wetland	189	6,728	74,632	291,676	266,904	148,007	115,061	59,102	18,234	629
Grassland	0	0	0	1,069	1,383	1,446	126	0	0	0
Open water	440	30,243	44,893	56,650	53,318	26,533	13,204	4,024	566	63
Agriculture	629	38,165	758,711	1,136,152	1,054,914	640,255	571,029	237,604	80,480	1,635
Extraction	9,620	152,597	112,735	129,334	107,264	66,144	58,977	35,273	12,323	63
Total Area	11,067	239,616	1,093,143	1,892,096	1,723,776	1,040,452	893,326	401,204	136,942	3,899
Area (%)	0.15	3.22	14.70	25.45	23.18	13.99	12.01	5.40	1.84	0.05

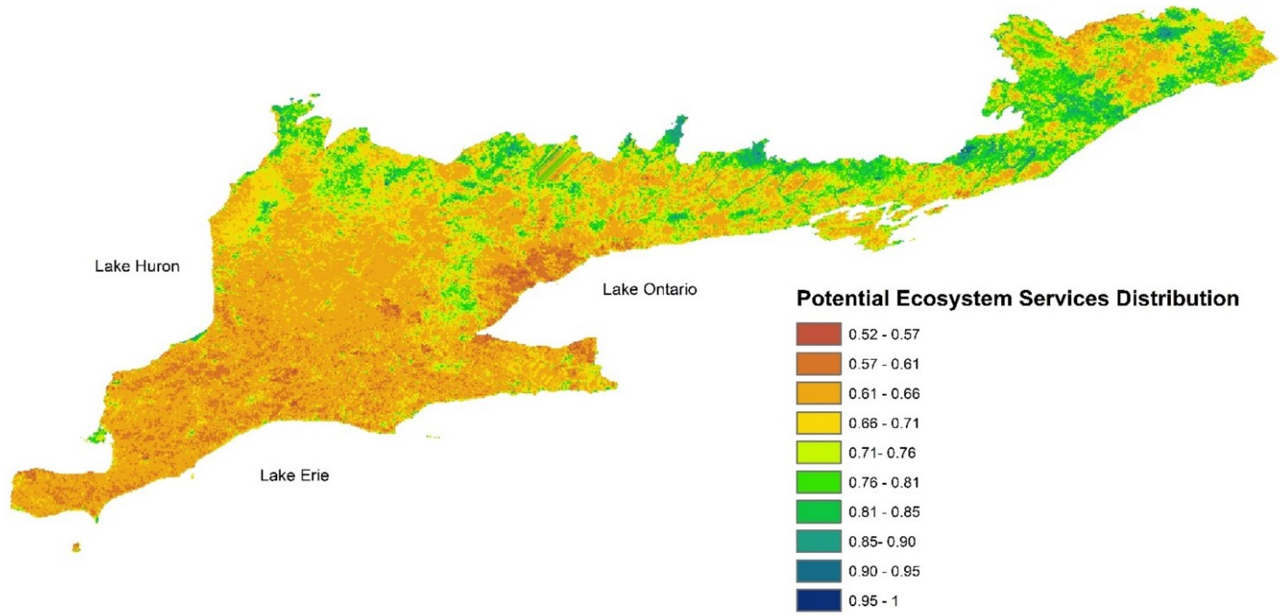

Figure 5.3: Distribution of potential ecosystem services in southern Ontario. The indices are rescaled between 0 and 1: 0 represents the minimum and 1 the maximum potential ecosystem services.

5.4.4 Modeling of realized ecosystem services

Finally, we used the Co$ting Nature model to map realized ES in southern Ontario. The indices are rescaled between 0 and 1, averaged into 10 indices, and named RS1 to RS10. These indices indicate that the lower (higher) the index, the lower (higher) the realized ecosystem services (Figure 5.4).

5.4.5 Valuation of realized ecosystem services

By overlaying the realized ecosystem services map on the land use map, we extracted the land use area under averaged indices from RS1 to RS10 (Table 5.3).

Using the equation (2), we calculated the total value of realized ecosystem services which came out to be $9.70 ±0.4 billion/year. This shows that the value of potential ecosystem services is approximately double of the value of realized ecosystem services (Figure 5.5).

5.5 Discussion

5.5.1 Similarity in the distribution map of unit values and potential ecosystem services

We valuated the selected region applying the value transfer method using unit values from regional studies. It is already demonstrated (in the paper 3) that the value transfer method yields better results when the unit values are taken from the regional studies rather than global studies. However, the primary purpose of this exercise was to show the difference in the values of potential and realized ecosystem services, and not the valuation itself. The potential ecosystem services represent the capacity of an ecosystem to generate services. Similarly, the unit value of an ecosystem for its ecosystem services describes its potential to supply ecosystem services.

The distribution map of potential ecosystem services, created by using Co$ting Nature model, is compared with the unit value distribution map to assess the accuracy of the model. The distribution map of potential ecosystem services shows that the natural land use areas have higher values on the potential ecosystem services index than the areas under agriculture. Similarly, the unit value distribution map for ecosystem services showed that value is higher for natural land use than agricultural land use (maps in supplementary material). Both of the maps, potential ecosystem service distribution map and unit value distribution map, show similar distribution patterns for southern Ontario. These patterns confirm that areas of high unit values hold higher potential for ecosystem services, and conversely, low unit values hold lower potential for ecosystem services. To further corroborate this claim, we created bar chart histograms of cells (pixels)

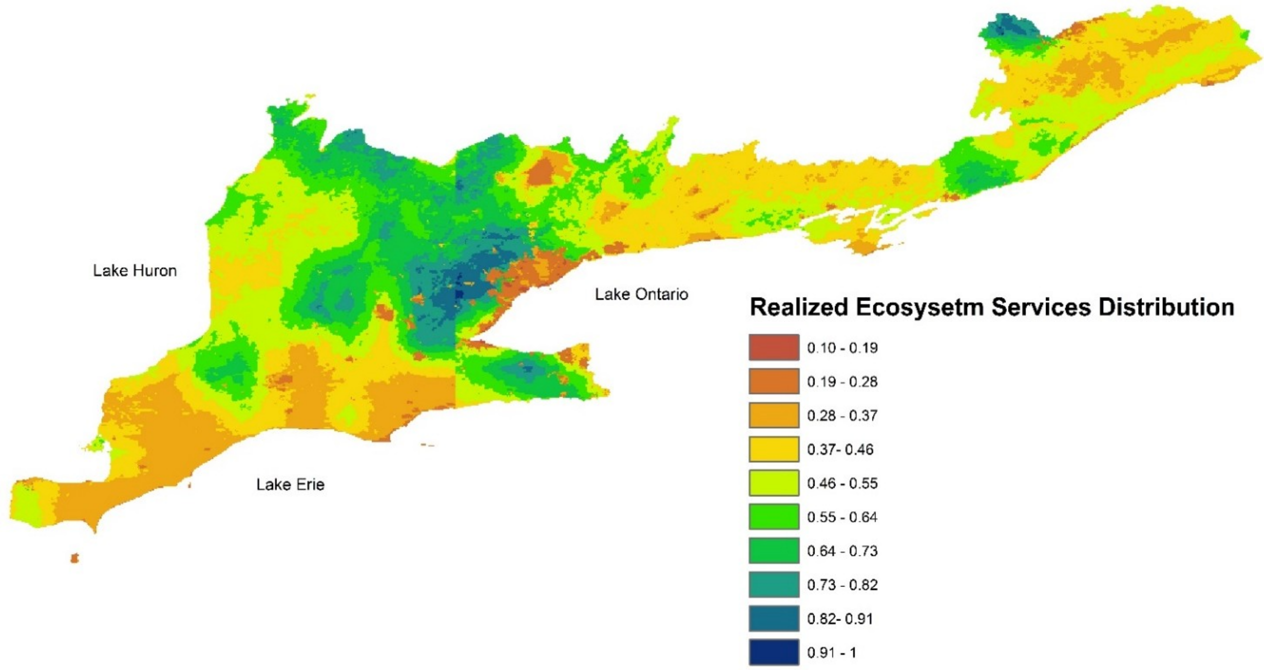

Figure 5.4: Distribution of realized ecosystem services in southern Ontario. The indices are rescaled between 0 and 1: 0 represents the minimum and 1 represents the maximum realized ecosystem services.

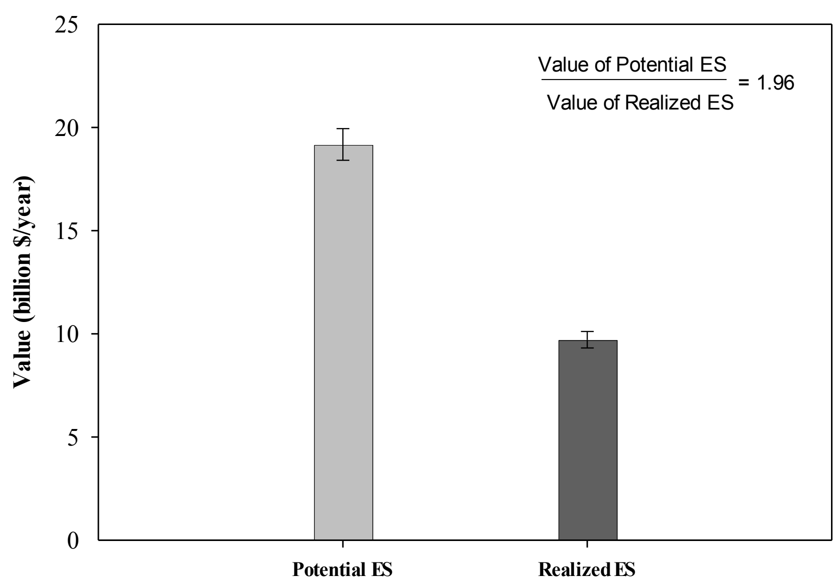

Figure 5.5: Total value (billion $/year) of potential and realized ecosystem services in southern Ontario.

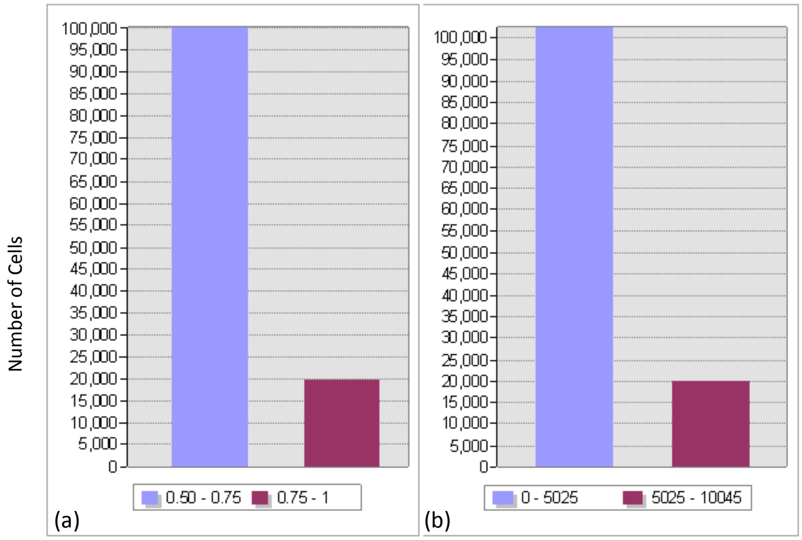

Figure 5.6: The number of cells that fall under a) low to medium (0.5-0.75), and medium to high (0.75-1) potential ecosystem services indices, and b) low to medium (0-5,025) and medium to high (5,025-10,045) unit value ($/ha/year) for the bundle of ecosystem services. The same number of cells for low to medium, and medium to high, indices and unit values (in both histograms) demonstrate that high unit values correspond to area under high indices and vice versa.

under low-medium and medium-high potential service indices and unit values for ecosystem services (Figure 5.6). The vertical bars in the first histogram represent the number of spatial cells under low to medium (0.5-0.75) and medium to high (0.75-1) indices for potential ecosystem services. Likewise, the vertical bars in the second histogram are created for the number of spatial cells under low to medium (0-5,025) and medium to high (5,025-10,045) unit values. Both histograms show an approximately equal number of cells (pixels) for low-medium and medium-high values for both unit values and potential service indices.

5.5.2 Distinction between realized and potential ecosystem services

In this valuation study, we used a model-based approach to differentiate the values of realized and potential ecosystem services in southern Ontario. In this approach, the Co$ting Nature model is partially used for its usefulness in capturing realized ecosystem services. This and other similar approaches can advance the field of ecosystem services in the areas which yet need answers. Accepting the edict “all models are wrong, but some are useful” (Box and Draper, 1987; Wainger and Mazzotta, 2011), partial use of imperfect models as interim products will help further the science of ecosystem services to make it more practical and robust.

The realized ecosystem services depend on potential ecosystem services, population density, and built infrastructure in the region. For example, water supply is modeled by the Co$ting Nature model based on clean water availability, population density and number of dams. As a result, the model calculates the realized ecosystem services indices by multiplying the potential indices by the normalised sum of all downstream users (Mulligan, 2015). Because realized ecosystem services are directly consumed by the people, their economic valuation can better illustrate the link between ecosystem services and human well-being. For this reason, realized ecosystem services are a portion of potential ecosystem services that matters to its beneficiaries (Wainger and Mazzotta, 2011). As a consequence, valuation of realized ecosystem services rather than potential ecosystem services will result in better-informed policy and decisions.

Burkhard et al. (2012) proposed an approach for the mapping of supply and demand of ecosystem services. They assigned an indicator to each land use type between 0 and 5 based on quantitative data, and expert judgement and knowledge of the landscape. The natural land use categories are assigned higher scores for supply of regulating, provisioning and cultural ecosystem services whereas built-up and urban areas are assigned higher values for demand of ecosystem services. However, this approach of quantification of ecosystem services faces a major problem regarding identification of appropriate indicators and lack of required data (Burkhard et al., 2012). On the other hand, we employed the Co$ting Nature model that uses global datasets to assign a relative index between 0 and 1 to each ecosystem based on its potential for and use of a bundle of ecosystem services. For realized ecosystem services, these indices are generated by assessing 117 maps of input data on distribution of population and infrastructure (Mulligan, 2015).

The selected region is modeled by using two tiles and then the distributions of realized ecosystem services for these tiles are combined to form a final map for southern Ontario. Our results demonstrate that the value of realized ecosystem services represents 50% of that of the potential ecosystem services in southern Ontario. A key factor responsible for the difference between potential and realized ecosystem services is the distribution of the population density (Turner et al., 2012). The distribution map of realized ecosystem services shows that the areas with higher indices have higher population density areas in their vicinity. Consequently, the projected increase in urban densification in southern Ontario (Ontario Ministry of Finanace, 2016) will further increase the value of realized ecosystem services. The realized ecosystem services mapping also highlights the areas where the potential for realized ecosystem services can be increased by investing in natural infrastructure.

The potential ecosystem services are valued using a constant unit value for a particular ecosystem, irrespective of its position on the landscape. However, this paper demonstrates that 'one size fits all' approach is not applicable for realized ecosystem services. The unit value of each ecosystem category is modified by applying the indices of realized ecosystem services, which are derived based on their spatial location (Eq. 5.2). The dominant factor affecting the modified unit values of ecosystems is their geographical position. For example, even though the unit value of forest for potential ecosystem services is 7.5 times higher than agriculture, the unit value of agriculture under RS10 is approximately equal to that of forest under RS1. Therefore, investing in natural infrastructure in the RS10 area will relatively increase the value of realized ecosystem services more than an equal investment in other areas (e.g., RS1, RS2 etc.).

Another major contribution of this paper is to show the relative importance of the location of ecosystem services, relative to the human benificiaries. A challenge in many western societies is to seek out balance between short-term private desires and long-term societal needs (Wallace, 2007). Therefore, the knowledge of the spatial distribution of realized ecosystem services' may help with land conversion and conservation decisions, by depicting not only the locations producing ecosystem services, but also how the flows of ecosystem services are channelled to nearby populations. In turn, this may inform where best to direct investment in natural land use to maximize its contribution towards human well-being.

In our analysis, we assess the economic value of a bundle of ecosystem services because the Co$ting Nature model can only model these ecosystem services. Despite this fact, the results of this model can by extrapolated to other ecosystem services by investigating the synergies between the modeled and rest of the ecosystem services. At the same time, the map of distribution of realized ecosystem services shows sharp change in the indices at the meeting point of two modeled tiles. This change is due to the fact that a large spatial extent is modeled, and therefore, this limitation will disappear with the selection of a smaller area that can be covered by one tile. Furthermore, our valuation methodology does not capture the financial and social situation of the beneficiaries. For example, the value of realized ecosystem services will be higher in the areas where poor or marginalized communities are totally dependent on these services. However, this dependency issue can only be addressed by conducting small-scale landscape studies.

5.6 Conclusions

In this study, we made a distinction between, and conducted monetary valuation of, potential and realized ecosystem services in southern Ontario. We draw out following conclusion from the study:

- The distribution maps of potential ecosystem services and unit values showed similar patterns for the selected region. Therefore, the unit values represent the capacity of ecosystems to generate services.
- The value of realized ecosystem services is about 50% of the value of potential ecosystem services in southern Ontario's landscape.
- The realized ecosystem service indices map can be used to identify the area for investment in natural infrastructure and payment for ecosystem services.
- Valuation of realized ecosystem services is a step towards meeting the growing need of decision makers for policy relevant research on ecosystem services and their contribution to society.

Chapter 6

Conclusions and further directions

Based on the outcomes of the research work presented in Chapters 2-5, I answered the research questions (RQ1-RQ4) and sub-questions (SQs) posed in Chapter 1. These answers are presented below in tabular form (Table 6.1).

Table 6.1: Answers to overall research questions (RQs) and sub-questions (SQs) in light of the research presented in this thesis (chapters 2-5).

Research Questions	**Answers**
RQ 1	I assessed the value of the Grand River watershed for four ecosystem services: water supply, water filtration, carbon sequestration and nutrient cycling. The analysis of four land use scenarios representative of years 1800, 1900, 2015 and 2050 shows that there is a decline in the combined value of the three non-consumptive ecosystem services from year 1800 to year 1900 due to deforestation and expansion in agricultural areas. The results indicate that increased anthropogenic activity initially degraded the natural ecosystems and subsequently degraded water, air and soil quality in the watershed. However, the value of these three non-consumptive ecosystem services increased in years 2015 and 2050 land use scenarios due to increased forest cover resulting from restoration and conservation practices in the watershed.
SQ 1.1	In this research work, I took into account the values of water filtration, carbon sequestration and nutrient cycling provided by the agricultural land use categories. The land use category with maximum sediment (SD_{max}) and phosphorus (PD_{max}) delivery rates is taken as the baseline, and other categories are valued relative to the baseline category. Thus, the unit value for the water filtration service for each land use category represents the relative sediment and phosphorus retention values. The carbon sequestration service provided by the agricultural land uses was valued based on the carbon storage in a stable soil organic pool (C_{is}) and on the carbon emission price recently implemented in Ontario. Further, I monetized the nutrient

	cycling (or the capacity of the soil to supply nutrients) based on the net nutrient uptake rates (supplied by the soil minus supplied by fertilizers) of the agricultural land uses and the local market price of elemental nutrients in the fertilizers. Given that carbon sequestration and nutrient cycling services depend on biomass, I accounted for variations in the biomass of agricultural land uses for past and future scenarios based on total factor productivity (TFP).
SQ 1.2	The value of water supply — a consumptive service — continuously increases from 1800 to 2050 scenarios due to the increase in population. By contrast, values of the three non-consumptive ecosystem services show a direct relationship to the size of natural land use in the watershed: the higher the natural land use, the higher the value of non-consumptive ecosystem services. I used a direct approach — market price method — for valuation of water supply based on the total water consumption and a wholesale rate for water supply in the watershed. Alternatively, the non-consumptive ecosystem services are valued by multiplying the unit values ($/ha/year) of these services for each land use with the corresponding land use area. I used different approaches for valuation of consumptive and non-consumptive ecosystem services, following the proposal that the ecosystem services with market values should not be valued using non-market valuation methods (Kuuluvainen, 2002).
SQ 1.3	I developed the local unit values for ecosystem services and, therefore, the value estimates are locally relevant. Further, I compared variations in the unit values (coefficient of variations) with other studies that use regional or global data. My results show that local datasets can significantly narrow down the uncertainties in the unit values.
RQ 2	Because of unique conditions and characteristics, wetland types need a framework for valuation of their ecosystem services that distinguishes each type. I put forth a framework that results in wetland value functions by integrating the magnitude of ecosystem services from wetlands with economic valuation. For the magnitude of an

	ecosystem service, a suitable indicator/ parameter needs to be selected to differentiate the level of service provided by different wetland types.
SQ 2.1	I developed water filtration value functions for four major wetland types in southern Ontario: bogs, fens, marshes and swamps. The key parameters used for sediment and phosphorus filtration are sediment and phosphorus accretion rates in the wetlands. I also performed a regression analysis to investigate the impact of wetland size on the sediment accretion rates. The results indicate that there are only weak correlations between sediment accretion rate and size for the different wetland types. Therefore, the values for water filtration service were determined using average accretion rates. Among all wetland types valued in this study, fens yield the lowest unit value, \$2750±2055/ha/year, while bog, marsh and swamp have 1.72, 2.66 and 1.56 times higher values
SQ 2.2	To offset the total phosphorus load generated by the conversion of all wetlands in southern Ontario to agriculture, I evaluated three alternatives: 1) best management practices (BMPs); 2) constructed wetlands (CWs); and 3) wastewater treatment plant (WWTP) upgrades. These alternatives cost (billion \$/year): 1) 13.60±5.35; 2) 2.90±1.90; and 3) 166.50±65.60, respectively. Though CWs are the cheapest among these alternatives, they would require 50% of the current wetland area for their construction.
RQ 3	Locally appropriate unit values and fine-resolution land use datasets can result in policy relevant estimates of ecosystem services. These datasets reduce uncertainty in the values and therefore increase the applicability of the value transfer method. The unit values derived at an earlier time may not be relevant due to advancements in science, growing understanding of complicated natural processes, availability of up-to-date knowledge on ecosystems, and changes in the economics of natural resources.
SQ 3.1	I assessed the validity of existing global unit value datasets, which are routinely being

	used for valuation of ecosystem services worldwide, by comparing value estimates with the regional and local datasets for the Grand River watershed, Ontario, Canada. The results show that global datasets led to inaccurate and inconsistent outcomes compared to the regional and local datasets. However, these datasets are easy to use and can therefore yield crude approximations of ecosystem services values in the absence of regional datasets. However, I recommend establishing regional datasets to improve the accuracy of value estimates.
SQ 3.2	The results of this study show that high-resolution land use data results in higher values compared to low-resolution data. Further, because of their higher unit values for ecosystem services, the resolution of natural land uses has a larger impact on the total value of the watershed.
RQ 4	I used an off-the-shelf tool to capture the distribution of potential and realized ecosystem services in terms of indices between 0 and 1. I rescaled these indices and devised a methodology to make realized ecosystem services indices amenable to monetary estimates. By applying this methodology to southern Ontario for the first time in the economic valuation literature, I am able to distinguish the values of potential and realized services.
SQ 4.1	The value of a bundle of six ecosystem services is calculated based on their use in southern Ontario. My results show that the value of realized ecosystem services is ~50% of the value of potential ecosystem services in this region. The value of realized ecosystem services depends on the population density and is, therefore, expected to increase in the future for this region.
SQ 4.2	Realized ecosystem services are the portion of services that matters to people; hence, their distribution can result in informed decisions regarding land use planning. Accordingly, a map of realized ecosystem services distribution generated from this study is significant for policy makers to locate the hotspots for investments in natural

	infrastructure. This realized ecosystem services map reflects a higher dependence of people on the areas with high index values, opposite to the potential ecosystem services map which is fully based on ecosystem types in an area. The map could be helpful in developing spatial conservation prioritization plans for the area.

6.1 Policy implications

The results of this work imply that, where possible, environmental policies should rely on unit values of ecosystem services derived from local data. The valuation of temporal land use scenarios (in chapter 2) helps illustrate the implications of human-environment interactions in monetary terms that may resonate with public stakeholders and inform local authorities. It may lead to a more general acceptance of the ecosystem services concept as a guiding principle in land use planning and (integrated) water management.

In addition to government agencies, many environmental nongovernmental organizations (NGOs, e.g., IUCN, WWF-Nature) (Berghöfer et al., 2016) and consultancy firms (e.g., Silvacom) are now carrying out economic valuation assessments of ecosystem services. The large uncertainties in these assessments could be significantly reduced by focusing more efforts on obtaining locally or regionally relevant unit values (as illustrated in chapters 2 and 4), and implementing novel frameworks for economic valuation (as illustrated in chapters 3 and 5).

Overall, I suggest that the policy relevance of ecosystem services valuation can be increased by taking following steps.

1. Ecosystem services values must reflect the local context to enhance their legitimacy, credibility and relevancy to policy-makers (Berghöfer et al., 2016).
2. There is a need to bridge the existing gap between academic ecosystem services valuation studies and policy making. The lack of interest in implementing ecosystem services in policy may partly be due to lack of awareness about what the concept entails. Thus, educating policy makers and

the public at large about ecosystem services and their valuation should be an important outreach goal of the environmental science community.

3 Ecosystem services valuation studies should be tailored to meet specific policy needs. For example, valuation of realized versus potential ecosystem services is a step towards making ecosystem services more tangible as a tool for land use planning.

6.2 Directions for further work

Based on the research work presented in this thesis, I suggest that the following research direction would go a long way in further improving monetary estimates, and achieve better land use planning.

6.2.1 Use of high-resolution land use data

Land use data is one of the key requirements for valuation of ecosystem services. I have shown (in Chapter 4) that a more detailed breakdown of land use categories helps to improve the economic estimates of ecosystem services. In addition to my results (presented in Chapter 4), other studies (e.g., Grafius et al., 2016) have also shown that fine resolution land use data, though it may be expensive to obtain, yields more accurate results. However, Grafius et al. (2016) suggest that land use resolution becomes less relevant when dealing with stock assessment models (e.g., carbon storage). Therefore, it is recommended to further investigate the relevance of land use resolution for a broader portfolio of ecosystem services.

6.2.2 Effects of landscape fragmentation on valuation of ecosystem services

With increasing human activity, natural ecosystems are degrading and disassembling into smaller pieces – called ecological fragmentation. This fragmentation may reduce the supply of ecosystem services and therefore has direct consequences for human well-being. In contrast, flows of some ecosystem services, such as carbon sequestration and storage, may be unaffected by fragmentation or even increase. Increased fragmentation decreases the supply of ecosystem services; however, the flows of ecosystem services can increase, decrease or stay constant. Therefore, the provision of ecosystem services can be affected positively, negatively, or remains insensitive to fragmentation (Mitchell et al., 2015). The economic valuation literature has partially addressed the effects of fragmentation on economic values of ecosystem services. For example, Brander (2011) analyzed the impacts of fragmentation resulting from transport infrastructure, but fragmentation caused by (sub)urbanization still remains largely unexplored. Hence, further research is needed to determine the effects of different types and mechanisms of fragmentation on the value of ecosystem services.

6.2.3 Development of wetland value functions

Wetlands are complex ecosystems and generate a plethora of ecosystem services. To assess all the ecosystem services from a wetland type, there is a need to develop wetland value functions for a complete suite of ecosystem services based on ecological indicators (e.g., trophic state) that should be conformable to economic valuation. These indicators can register a change in the magnitude of flow of ecosystem services from a wetland based on its ecological state. In fact, it will introduce another type of unit values which will involve biophysical units of local ecosystems and local market values. As a result, valuation estimates based on these unit values will be locally appropriate and policy-relevant. Furthermore, the idea of developing value functions can be expanded to include other ecosystem types.

6.2.4 Exploring synergies and trade-offs among ecosystem services

Agroecosystems are the main food providers and cover a significant portion (over 40%) of the continental surface; they also reduce ecosystem services provided by natural ecosystems (e.g., pollination, water quality). Typically, agrocecosystems are considered as producers of provisioning services (e.g., food, fiber), but recent studies have recognized that they provide other ecosystem services as well, such as carbon sequestration, water quality, and cultural services (Power, 2010). In Chapter 2, for example, I have quantified the role of agricultural land uses in carbon sequestration, nutrient cycling and water quality services. These services must be balanced against the disservices of agroecosystems compared to natural ecosystems, such as biodiversity loss, nutrient and sediment loads to water bodies, and emission of greenhouse gases (Power, 2010). Due to their key role in food security, agricultural land uses are and will remain an integral part of our landscapes. Thus, there is an urgent need to explore synergies (where an increase in one service increases the other) and tradeoffs (where an increase in one service decreases the other) between ecosystem services generated from natural and agricultural ecosystems.

Ecosystem services are dependent on each other, and relationships among them can be non-linear. Consequently, increasing agricultural ecosystem services results in trade-offs and minimizing these trade-offs will be a step towards achieving more sustainable landscapes. This research can be further advanced by forming bundles of ecosystem services with a similar response to land use changes. Additionally, if numerical or functional relationships are formed between ecosystem services, it can make it possible to valuate more ecosystem services with a smaller budget and less time. Synergies and tradeoffs arise from the land use changes (Deng et al., 2016) and therefore can better inform the decisions related to land use management. At the same time, valuation is a hopeful avenue for carrying out synergy and trade-off analyses where several ecosystem services, netting multiple dimensions and measured in different biophysical units, are involved.

6.2.5 Linear optimization model for land use design

The supply and demand of ecosystem services depend on land use planning and management. Ecosystem services can be a useful and practical tool to guide multi-objective land use planning. In the face of climate change, ever-increasing population, and ecosystems degradation, there is an overarching need to design optimal landscapes using integrated approaches. In my opinion, taking into account potential and realized values of all ecosystem services can help achieve this goal (Figure 6.1).

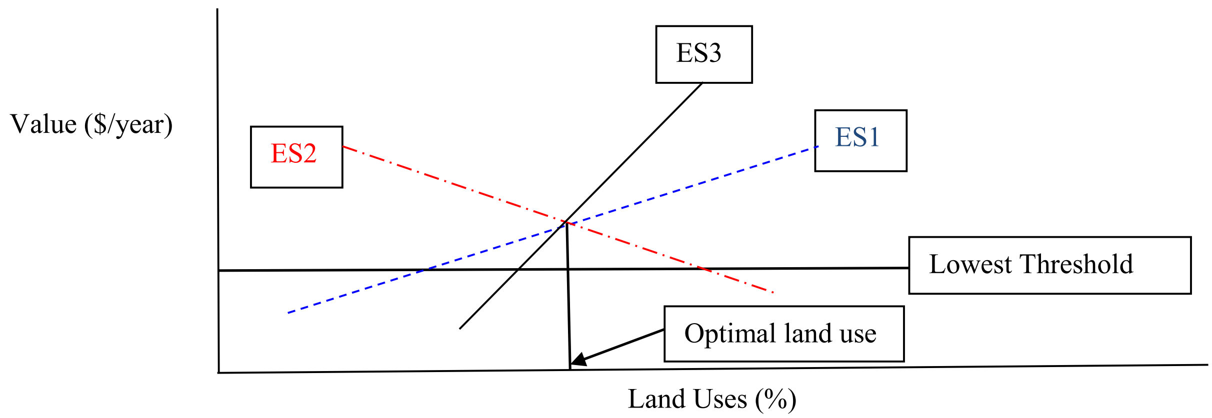

Figure 6.1: The use of ecosystem services framework for optimization of watershed land use. ES1, ES2, ES3 represent three different ecosystem services 1, 2 and 3, respectively. The optimal land use will ensure the values of different ecosystem services above the lowest threshold.

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Appendix A
Supplementary Material: Chapter 2

Land use for year 1800

In the map of year 1800 taken from Historical Atlas of Canada, each dot represents 200 people and there are 13 dots in total along the Grand River watershed (Harris, 1997). Therefore, the total population in the watershed for year 1800 is calculated as 13*200=2600 people.

Using updated files sent by Sadia Butt (through personal communication) for the pre-European land use in southern Ontario, Grand River watershed data is clipped and a map is created for Grand River watershed (Figure AA1). There is 20.46% missing data within this watershed. The other major land uses are extrapolated with the existing proportion to accommodate this 20.46% of missing land use. The new percentages of major land uses in the watershed are: forest (conifer) is 0.95%, forest (deciduous) is 76.92%, forest (mixed) is 6.06%, cleared land is 0.89% and wetlands are 15.18%. Cleared land is distributed among residential and agricultural uses. The population of the watershed in year 1800 was 2600 (Historical atlas of Canada); therefore the residential area is allocated relative to the current population requirements. The rest of cleared land is assigned to agriculture keeping relative change in sub-categories proportional to current agricultural land use (2015) sub-categories in the watershed. The final areas of different land uses are given in Table 2.2 (of the main text).

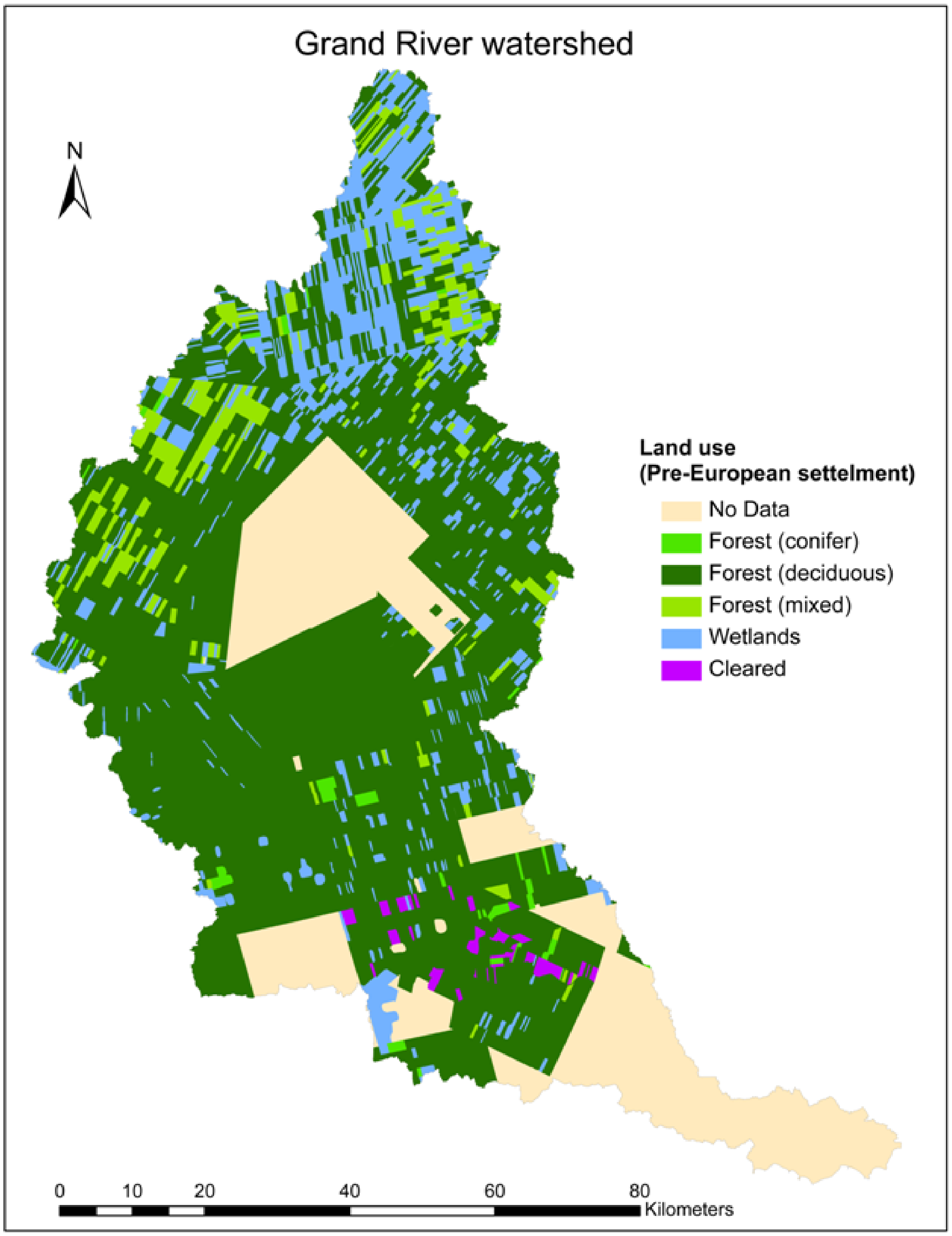

Figure AA 1: Pre-European land use for the Grand River watershed

Sediment and phosphorus delivery rates

The soil erosion rates vary from minimum of 0.90 t/ha/year for CC (continuous corn) rotation and RIPL (Ridge Planting) tillage system to maximum of 2.91 t/ha/year for CS (alternating corn and soybeans) rotation and FPL (fall moldboard ploughing) tillage system in Big Creek watershed, from minimum of 1.57 t/ha/yr for CC (continuous corn) rotation and RIPL (Ridge Planting) tillage system to maximum of 4.48 t/ha/yr for CC (continuous corn) rotation and FPL (fall moldboard ploughing) tillage system in Newbiggen creek and from minimum of 1.74 t/ha/year for 3C3A (alternating 3 years corn and 3 years alfalfa) rotation and NT (No-till) tillage system to maximum of 7.00 t/ha/year for 2C2A (alternating 2 years corn and 2 years alfalfa) rotation and FPL (fall moldboard ploughing) tillage system in Avon/Stratford watershed. The average sediment delivery rate from continuous corn is 0.37±0.24 t/ha-year but from alternating corn and soybean is 0.46±0.28 t/ha-yr. Therefore the sediment delivery rate for soybean is calculated as (4*(0.46±0.28)- 2*(0.37±0.24))/2= 0.55±0.445 t/ha/yr. Similarly the delivery rate for alfalfa is calculated as (4*(0.46±0.29)- 2*(0.37±0.24))/2=0.55±0.45 t/ha-yr.

Plants and vegetation improves the soil structure, reduce sediment erosion and filters out chemicals from the water (Belcher et al., 2001; Wall et al., 1995; Elmore and Beschta, 1987). Rates of soil erosion are highly variable and depend on the soil type, slope and cultivation practices. The different types of vegetations and forest can be analyzed for their erosion rates based on USLE (universal soil loss) equation. The C factor in the equation reflects the effect of vegetation on soil erosion (Yan et al., 2003).

Shaver et al. (1994) gave a range of typical export rates (minimum, maximum and median) of total suspend solid (TSS), and total phosphorus (TP) from different land covers based on data collected in the Pacific Northwest (PNW). These rates for phosphorus can vary considerably due to variation in the concentrations, land use and other regional factors (climate). The grass and pastures have almost the same rate of loading for TSS and TP.

According to van Vliet et al., 1978, the dominant source of fluvial sediment in southern Ontario watersheds is cropland (70-100%) while stream banks and channel erosion is a minor source (0-30%). The three prediction models are used for regions under horticultural crops (potatoes, tomatoes) with high sheet erosion. The sediment delivery ratio curve constructed for Canadian agricultural watersheds is 10%

lower than similar SCS (Soil Conservation Service) developed curve for U.S. The universal soil loss equation (A = R*K*L*S*C*P) is used for soil erosion losses from various crops (Vliet et al., 1978).

The average and range of erosion losses for different crops given by van Vliet et al., (1978) are used to compute delivery rates for different land covers. The average delivery ratio of 18.36% is used based on average value of drainage basin series (Roehl, 1962; Vliet et al., 1978) which is 16.36% and modified value for predominant watershed soil texture (Vliet et al., 1978) which is 20.36% for 11 agricultural watersheds in southern Ontario.

Similarly, the phosphorus delivery rates are taken from a number of studies and mean (±SD) are presented in Table AA1.

Table AA 1: Sediment and phosphorus delivery rates for different land uses from local and regional studies.

Land Cover	Sediment delivery rates (ton/ha/year)	Land Cover	Phosphorus delivery rates (Kg/ha/year)
based on Fox and Dixon, 1990		**Hore et al., 1998**	
Corn	0.37±0.24	Corn (rotation)/row crop	0.18±0.08
Soybean	0.55±0.445	Oat	0.13
Alfalfa	0.55±0.45	Alfalfa	0.14±0.06
Shaver et al., 1994		Corn (continuous)/ row crop	0.28±0.02
Forest	0.09	**Shaver et al., 1994**	
Pasture	0.34	Forest	0.11±0.02
Grass	0.35	Pasture	0.13±0.12
Urban area (residential + commercial)	0.44	Grass	0.13±0.12
based on Vliet et al., 1978		Urban area (residential + commercial)	0.675±0.13
Horticultural crops (tomatoes + potatoes)	1.71±0.52	**Karst-Riddoch et al., 2014**	
Beans	1.40±0.39	Cropland	0.28±0.08
Continuous corn	1.30±0.81	Forest	0.07±0.03
Corn in rotation	0.70±0.55	Hay-pasture	0.10±0.04
Tobacco	0.64±0.26	Wetland	0.05±0.02
Small grains	0.72±0.50	Open Water	0.26
Meadow in rotation	0.52±0.38	**modified from Winter, 1998**	
Permanent pasture	0.08±0.06	Urban	0.5±0.04
Woodlands/forest	0.04±0.03	Pasture	0.5±0.04
Fernandez et al., 2003		Row Crops	0.3±0.02
Forest	0.18	Non-row crops	0.25±0.02
Grassland	0.62	Woodland/Forest	0.1±0.01
Crop	6.6	Atmospheric/Open water	0.37±0.03
Mahmoudzadeh et al., 2006		**Donahue., 2013**	
Forest	0.8	Forest	0.2±0.06
Pasture	2.2	Non-row crops	0.96±0.44
Crop	3.1	Pasture	1.22±1.08
		Mixed agriculture	0.9±0.32
		Urban	1.5±0.7

Carbon Sequestration

For forests, Gale et al. (2009) assumed that root biomass represents 20% of the above ground biomass (Nogueira et al., 2014) and total plant carbon is assumed to corresponds on average to 50% of the total living biomass which is average proportion of carbon in dry plant biomass (typical range: 43.4-55.6% for temperate species as reported by Thomas & Martin, 2012). Gale et al. (2009) used measurements with a LECO CR12 Carbon Analyser to estimate the carbon sequestration by organic soil. Carbon lost through soil respiration was measured and subtracted from sequestration by soil and biomass. Carbon sequestration rates representative of local forests are used in our study, based on the data from Gale et al. (2009) for Wellington County within the Grand River watershed.

The deciduous forest is dominantly deciduous and represents 85% of Sugar Maple (deciduous), 7.5% of Norway Spruce (coniferous) and 7.5% of White Pine (coniferous). Coniferous is composed of 15 % of Sugar Maple (deciduous), 42.5% of Norway Spruce (coniferous) and 42.5% of White Pine (coniferous) while mixed forest made up of 42.5% of Sugar Maple (deciduous), 42.5% of Norway Spruce (coniferous) and 15% of White Pine.

Carbon sequestration rates for local species were calculated as 96 tCO_2 ha^{-1} yr^{-1} for Sugar Maple (deciduous); 22 tCO_2 ha^{-1} yr^{-1} for White Pine (coniferous); and 25 tCO_2 ha^{-1} yr^{-1} for Norway Spruce (coniferous). The deciduous forest rates are (0.85*96+0.075*22+0.075*25= 85 t CO_2 ha^{-1} yr^{-1}), for coniferous (0.15*96+0.425*22+0.425*25= 34.3 tCO_2 ha^{-1} yr^{-1}) and for mixed (0.425*96+0.15*22+0.425*25=54.7 tCO_2 ha^{-1} yr^{-1}). (Rates are also confirmed through personal communication with Nigel Gale).

While the traditional models applied to temperate tree species give the carbon sequestration rate of 35.17 (±5.75) t C ha-1 yr-1 (128.96±21 tonne of CO_2 ha^{-1} $year^{-1}$) for Sugar Maple; 15.080 (±2.72) t C ha^{-1} yr^{-1} (55.29±10 tonne of CO_2 ha^{-1} $year^{-1}$) for White Pine, and 30.121 (±3.47) t C ha^{-1} yr^{-1} (110.44±12.72 tonne of CO_2 ha^{-1} $year^{-1}$) for Norway Spruce (Gale et al., 2009; Alemdag, 1983; Alemdag, 1984; Ter-Mikaelian & Korsukhin, 1997; Perala & Alban, 1994; Jokela et al., 1986).

The root biomass density for forest was estimated from the existing data from literature for root biomass. Cairns et al., 1997 formed relationship between root biomass density and root:shoot ratios. The relationship was statistically tested. The linear regression analysis showed that important predictors of root biomass density (RBD) were above ground biomass density (ABD), age and latitudinal category (tropical, temperate and boreal) and were responsible for 84% of the variation in RBD. These estimates of root biomass density (RBD) were 20% higher when compared with the generalized root:shoot ratios (R/S) for forests in the United States (Cairns et al., 1997). The relationship below showed that root biomass density (RBD) is 20% of the above ground biomass density (ABD).

The Gale., 2009 assumed that carbon content is 50% of total dry mass. However, we applied the variation to the C content which varies from 43.4 to 55.6% of total biomass for temperate species (taken from Thomas & Martin, 2012).

The carbon fixation by the crops is estimated by Winans et al., 2015 for high and low yield based on C_P (product biomass in harvested plant that is equal to Yield*C content), Cs (carbon fixed in stubble residue including straw and litterfall), C_R (C in root biomass), C_E (C in root turnover and root exudates), Ci(carbon input to soil) and Cis (C storage in a stable soil organic). The net primary production represents the gain of C in a system and is calculated as NPP= C_P + Cs + C_R + C_E (Winans, et al., 2015)

Here, we used average value of CO_2 sequestration for low (6.5 Mg ha^{-1} $year^{-1}$) and high yield (9.3 Mg ha^{-1} $year^{-1}$) of grain corn and that is 53.85 tonne of CO_2/ha/year [NPP is ((7.5+10.73)/2 + (c to soil is (4.58+6.55)/2)*44/12] (Winans et al., 2015). But, we took only carbon fixed in the NPP ((7.51+10.73)/2)*44/12 that is 33.44 tonne of CO_2/ha/year for an average yield of 7.9 Mg ha^{-1} yr^{-1} (7.9=(6.5+9.3)/2), because the C input to soil is coming from the C in NPP (plant fraction, carbon in the roots, in the litterfall and root exudates) and variation for low yield (5.2 Mg ha^{-1} yr^{-1}) and high yield (8 Mg ha-1 yr-1) . The emission of CO_2 from corn field due to fuel, fertilizers and chemical consumption is estimated by Belcher et al., 2001 as 3.941 tonne of CO_2/ha/year (Belcher et al., 2001). So the net sequestration for grain corn is 33.44-3.941=29.491 tonne of CO_2 /ha/year. The average yield gives maximum and minimum yield with 20% variation (7.9+7.9*0.2=9.5 or 7.9-7.9*0.2=6.32) so the CO_2 sequestration rate is varied by 20% for maximum and minimum value for corn. Therefore the calculated rate of CO_2 sequestration for corn is 29.491 ±5.90 tonnes of CO_2 per hectare per year.

For hay, the average carbon gain in the system 23.05 tonne of CO_2 ha^{-1} $year^{-1}$ (23.05=(18.18+27.92)/2) for an average yield of 6.6 Mg ha^{-1} yr^{-1} (6.6=(5.2+8)/2)(Winans et al., 2015). The CO_2 emissions from fuel, fertilizers and chemicals used for hay are 0.095 tonne of CO_2 ha^{-1} $year^{-1}$ (Belcher et al., 2001). Therefore the net carbon sequestration rate of hay is 23.05-0.095=22.955 tonne of CO_2 ha^{-1} $year^{-1}$. Specifically, for soybean the emission is 0.83 tonne of CO_2 ha^{-1} $year^{-1}$ so the sequestration rate is 23.05-0.83= 22.22 tonne of CO_2 ha^{-1} $year^{-1}$. The variation of maximum and minimum yield to average yield is 20%. There the CO_2 sequestration rate is varied by 20% and CO_2 sequestration rates for hay and soybean are 23.05±4.61 and 22.22±4.44. The value of NPP for wheat is varies from 685, 705, 648 and 544 $g/m^2/yr$ (Stellacci & Caliandro; 2007) under different cropping systems (W1b wheat-bean double-crop included in the three-course rotation; W2 = wheat included in the three -course rotation; Wb = continuous double cropping of wheat/bean; CW = continuous wheat). This comes out to be 6.85, 7.05,6.48 and 5.44 Mg/ha/year with average and standard deviation of 6.45±0.72 Mg/ha/year (Stellacci & Caliandro; 2007). It gives the CO_2 sequestration rate of 23.67±2.64 t CO_2/ha/year. Subtracting the CO_2 emission value of soybean which is 0.83, the net CO_2 sequestration by wheat is 22.84±2.64 t CO_2/ha/year.

For wetlands, Gower et al., 1997 calculated the above ground net primary production (ANPP) for mature forest stands ranging from 1.170 Mg C ha^{-1} yr^{-1} (4.29 tonne CO_2 ha^{-1} $year^{-1}$) to 3.520 Mg C ha^{-1} yr^{-1} (12.90 tonne CO_2 ha^{-1} $year^{-1}$). The average net primary production of boreal deciduous and coniferous forests was calculated as 3.6 Mg C ha^{-1} yr^{-1} (13.20 tonne CO_2 ha^{-1} $year^{-1}$) and 1.4 Mg C ha^{-1} yr^{-1} (5.13 tonne CO_2 ha^{-1} $year^{-1}$) respectively. The total above and below ground net primary production values estimated from the field measurements in Canada, US, China, Sweden and Finland are 4.7 Mg C ha^{-1} yr^{-1} (17.23 tonne CO_2 ha^{-1} $year^{-1}$) for deciduous broad leaves forest and 2.7 Mg C ha^{-1} yr^{-1} (9.9 tonne CO_2 ha^{-1} $year^{-1}$) for coniferous forest (Gower et al., 1997; Bhatti & Tarnocai., 2009).The NPP values based on the C budget model of the Canadian Forest Sector (CBM-CFS2), vary from 1.5 to 3.9 Mg C ha^{-1} yr^{-1} (5.5 to 14.3 tonne CO_2 ha^{-1} $year^{-1}$) based on forest type, age, location and productivity (Bhatti & Tarnocai., 2009; Li et al., 2003)

The total NPP reported for fens and bogs in Western Canada is 5.1 Mg C ha^{-1} yr^{-1} (18.7 tonne CO_2 ha^{-1} $year^{-1}$) with 3.4 Mg C ha^{-1} yr^{-1} (12.5 tonne CO_2 ha^{-1} $year^{-1}$) above ground and 1.7 Mg C ha^{-1} yr^{-1} (6.2 tonne CO_2 ha^{-1} $year^{-1}$) below ground components (Bhatti & Tarnocai., 2009; Campbell et al., 2000). But the shrubby swamp and marshes can produce more biomass through net primary production as compared to peatlands (bogs and fesn) (Vitt et al., 2001). Vitt et al., 2000 previously estimated that fens and bogs over long terms (1000 years) are net carbon sink of 19.4 gC m^{-2} yr^{-1} (0.71 tonne CO_2 ha-1 $year^{-1}$)

According to Joosten and Clarke (2002), pristine fens remove 2.5 Mg ha^{-1} yr^{-1} (CO_2) and release 2.97 Mg ha^{-1} yr^{-1} (as CH4) bogs remove 3.1 Mg ha^{-1} yr^{-1} and release 0.53 Mg ha^{-1} yr^{-1}. It makes the fens as C source and bogs as C sink. The total sequestration of 5.6 Mg ha^{-1} yr^{-1} (2.5+3.1) and release of 3.5 Mg ha^{-1} yr^{-1} (2.97+0.53) shows that peatlands are C sinks (Bhatti & Tarnocai., 2009; Joosten & Clarke., 2002). The above ground net primary production is used to measure the total net primary production in the wetlands. The NPP variation is equally sensitive to changes in vegetation types (moss, shrubs, woody) in a wetland and among different types of wetlands (swamps, bogs, marshes). The below ground NPP is considered 50 % and 30% of the above-ground NPP for peatlands (fens and bogs) and for non-peat accumulating wetlands (swamps and marshes), based on literature review (Campbell., et al., 2000; Vitt et al., 2001).

The carbon sequestration rates are based on the above and below ground biomass in different types of wetlands (fens, bogs, marshes, swamps). The amount of NPP variation highly depends on the vegetation type in a wetland and on different wetland types (Campbell., et al., 2000; Vitt et al., 2001). The mean above ground NPP ± SD for the fen and bogs is 337±142 g m^{-2} yr^{-1}(12 ± 5 tonne CO_2 ha^{-1} yr^{-1}) and for marshes and swamps is 924±463 g m^{-2} yr^{-1} (34± 17 tonne CO_2 ha^{-1} yr^{-1}). The mean total NPP ± SD for fens and bogs is 506 g m^{-2} yr^{-1} [337+0.5*337=506, adding 50% for below ground biomass] (19 tonne CO_2 ha^{-1} yr^{-1}) and for marshes and swamps is 1201 g m^{-2} yr^{-1} [924+0.3*924=1201, adding 30% for below ground biomass] (44 tonne CO_2 ha^{-1} yr^{-1}) (Vitt et al., 2001). Campbell et al., 2000 measured the NPP at different sites and then calculated the pooled mean for different sites. The average total net NPP for wetlands can be calculated as (19+44)/2= 31.5±15.75 tonnes CO_2 ha^{-1} yr^{-1} (because the above ground biomass variation is 50% so the same is assumed for overall (above + below) value). In the Grand River watershed swamps are 89.40%, marshes are 9.75 %, fens 0.15% and bogs are 0.70% (SOLRIS, 2015), so I have increased the value of CO_2 sequestration by 0.99*(44±17)+0.01*(19±5)=44±17 CO_2 ha^{-1} yr^{-1}.

Silvipasture is the area where trees are combined with the pasture and this area has increased potential for CO2 sequestration. In Thevathasan et al., (2004) study, the silvipasture has tree density of 111 trees per hectare because it is similar to most of operational silvipastoral systems (Thevathasan and Gordon, 2004). The carbon sequestration quantification for above and below ground biomass was based on the destructive sampling of trees. Above and below ground biomass production of grass was quantified by manual harvesting. For net carbon sequestration, the soil respiration rates were taken into account. The total C sequestered per tree by hybrid poplar, Norway spruce and grass is 125.9±16.3 (kg) and 49.8±8.4 (kg) respectively. The net carbon sequestered by grass is 196.6 ± 80.9 g/m^2 (719±296 CO_2 g/m^2 or 7.19±2.96 t CO_2/ha/year). Assuming the tree density of 111 trees per hectares, the average total CO_2 sequestration is 111*((125.9±16.3)+(49.8±8.4))/2 and converting kg to tonne +(7.19±2.96); the final CO_2 sequestration rates are = (9.75±1.02)+(7.19±2.96)= 16.94±3.13 t CO_2/ha/year.

Nutrient price

Using the rates of anhydrous ammonia (82-0-0), urea 46%, nitrogen solution (UAN) 28%, and ammonium nitrate 34% from Ontario farm input monitoring survey (McEwan, 2015), the values of 1 Kg nitrogen are $1.17, $1.26, $1.38 and $1.66, respectively. The average value of 1Kg of nitrogen is $1.37/KgN (close to $1.357/Kg N used in field crop budget 2016, OMAFRA).

Using the rates for mono-ammonium phosphate 11-52-0, Di-Ammonium Phosphate 18-46-0, Triple Superphosphate 0-46-0 and Nitrogen rate of $1.37/Kg, the values of 1 Kg of P_2O_5 are $1.094 ($2.50/Kg P), $1.11 ($2.54/Kg P) and $1.65 ($3.78/Kg P) respectively with an average value of $1.28/Kg P_2O_5 ($2.94/Kg P). The calculation of elemental P is based on ratio of P (30.97) to P_2O_5 (141.94)=(30.97*2/141.94=0.437). For -Ammonium Phosphate 11-52-0 (110Kg N and 520Kg P_2O_5 in one tonne), the value is ($719.85-110*$1.37) =$569.15 and then 520*0.437=227.24 Kg of P. The value for 1 Kg of P is $569.15/227.24=$2.50/Kg P. Using the Muriate of potash (60%), the value of 1 Kg of K_2O is $ 0.93/Kg. The value of 1Kg of K is $1.12/Kg K. Applying the rates of $1.37/Kg N, $2.94/Kg P, and $1.12/Kg K, unit values of nutrient cycling for different land uses are obtained.

Sensitivity analysis on land use

There are different land use datasets for the Grand River watershed (GRCA, 1999; SOLRIS, 2015; AAFC, 2010). For our 2015 baseline, we rely on the 1999 GRCA data, updated with wetland information for 2015. As such, we do not account for the expansion of the urban area over the last 16 years. In order to test the effect of land use variability due to the uncertainty in the data sets, plus the urban area changes, we perform a sensitivity analysis for different land uses and compare the results with our baseline estimates. We increase land use area for each category by 10% and 50%, and reduce all other land uses proportionally. Additionally, we increase each land use by 10% without reducing the others.

For the three non-consumptive ecosystem services, the effects of change in land use area are translated into a percent change in the total value for the watershed (Table 2.9). The results of the sensitivity analysis show that variations in the area of any of the land uses results in relatively small changes in the whole-watershed value of ecosystem services.

Sensitivity analysis on TFP

A second sensitivity analysis was performed for different values of TFP growth. The TFP growth rate for the period of 1871-1921is linearly extrapolated for the period of 1800-1940 (see section 2.5.2) and new biomass yield was used for valuation of pre-European land use (1800). Considering that there is no growth between 1800 and 1871, we used the TFP rate of 1871 (which is 5.12= ($1.0121^{69}*1.0114^{6}*1.0082^{69}$)) for pre-European land use and obtained a new value which shows an increase of 0.04% in the previous value (based on extrapolation) of the watershed.

For the target land use (2050), we increased the biomass yield by applying a factor based on the 1990-2009 TFP growth rate which is 1.14 (Darku et al., 2016). The incremental factor based on this TFP was 1.49 (1.0114^{35}) and the total value of the watershed was obtained as \$625±37 million/year. Based on global agriculture TFP, the recommended rate is 1.75% to meet the food demands of a growing world population (Zeigler and Steensland, 2015) and the incremental factor is 1.84 (=1.0175^{35}). Based on this new increment factor, the value (\$631±42 million/year) shows a 0.96 % increase in the previous value (\$625±37 million/year) of the watershed.

Appendix B

Supplementary Material: Chapter 3

Regression models for wetlands

Regression models for different wetlands (presented in Figure 3.4) are based on extensive research for data for different wetland types presented in tables AB1-AB4.

Table AB 1: Sediment accretion rates for bogs.

Bog name	Area	Accretion rate TSS (cm/yr)	Reference
Wylde Lake Bog	460.72 ha (cwi)	0.059±0.001*	(Shiller, 2013)
Marcell S-2 Bog	3.2 ha	0.24	(Wieder et al., 1994)
Big Run Bog	15ha	0.31	(Wieder et al., 1994)
Tub Run Bog	23 ha	0.23	(Wieder et al., 1994)
Cranberry Bog 1	65 ha	0.055	(Kadlec and Robbins, 1984)
Cranberry Bog 2	65 ha	0.23	(Kadlec and Robbins, 1984)
Alfred Bog	4000	0.05*	Bird and Hale Limited, 1984
Burns Bog	4000 ha	0.42	(Biggs, 1976)
Sifton Bog	41.6 ha	0.18	(Le Roux and Marshall, 2011)
Mer Bleue Bog	2800 ha	0.21	(Talbot et al., 2010)

Wylde Lake Bog is located in Luther marsh, Grand River watershed and area is calculated from Canadian wetland inventory (cwi). *Results are calculated for long time period (more than 300 years) and will be discarded.

Table AB 2: Sediment accretion rates for fens.

Fen name	Area	Accretion rate TSS (cm/yr)	Reference
Drosera Fen, Yosemite National Park	5.03 ha	0.39±0.15	(Drexler et al., 2015)
Porcupine Fen, Yosemite National Park	0.98 ha	0.16±0.02	(Drexler et al., 2015)
Kiln Fen, Sagehen basin	2.2 ha	0.08±0.04	(Bartolome et al., 1990)
Two field East Fen	0.8 ha	0.05±0.009	(Bartolome et al., 1990)
West Fen	0.1 ha	0.03±0.02	(Bartolome et al., 1990)
Bagno Bruch	39 ha	0.13	(Fia kiewicz-Kozie et al., 2014)
Bagno Mikołeska	5 ha	0.16	(Fia kiewicz-Kozie et al., 2014)
Abeille fen	3.5	0.15	(Van Bellen et al., 2013)
LG1 fen, Quebec	20	0.12	(Beaulieu-Audy et al., 2009)

Table AB 3: Sediment accretion rates for marshes.

Marsh name	Area (ha)	Accretion rate TSS (cm/yr)	Reference
Hank's marsh	438.84	0.28±0.03*	(Graham et al., 2005)
Upper Klamath NWR	3484	0.54	(Graham et al., 2005)
Squaw Point	133	0.42±0.03	(Graham et al., 2005)
Corte Madera Marsh	121	0.4±0.07	(Callaway et al., 2013)
Barataria basin marsh	4780	0.65	(Hatton et al., 1983)
Dyke Marsh	37.5	0.31	(Elmore et al., 2015)
Sweet Hall marsh	401	0.53±0.11	(Neubauer et al., 2002)
Great Marsh, Delaware	6880[§]	0.5	(Church et al., 1987)
Ogeechee marsh, Georgia, USA	700	0.21	(Loomis and Craft, 2010)
Altamaha marsh	3700	0.12	(Loomis and Craft, 2010)
Satilla marsh	1700	0.23	(Loomis and Craft, 2010)
Jug Bay marsh Maryland	607[1]	0.5	(Khan and Brush, 1994)
Gleason marsh	85[§]	0.27	(Darke and Megonigal, 2003)
Walkerton marsh	16[§]	0.12	(Darke and Megonigal, 2003)

*Rates are average of ^{210}Pb and ^{137}Cs models

[1]http://dnr2.maryland.gov/wildlife/Documents/NaturalAreas/JugBay.pdf (Department of natural resources Maryland)

[§] Area taken from US national Inventory

Table AB 4: Sediment accretion rates for swamps.

Swamp name	Area	Accretion rate TSS (cm/yr)	Reference
Tamarack swamp	1618 ha	0.14	(Wieder et al., 1994)
Cranesville Swamp	809 ha	0.19	(Wieder et al., 1994)
Black swamp Arkansas	1804 ha§	0.28	(Hupp and Morris, 1990)
Walden swamp	26 ha§	1.26	(Meadowlands Environmental Research Institute, 2011)
Eight Day swamp	7.85 ha§	0.83	(Meadowlands Environmental Research Institute, 2011)
Backswamp, Alabama	1163 ha§	0.5 ± 0.1	(Kidd et al., 2015)
Okefenokee Swamp		0.08	(Craft et al., 2008)
Louisiana swamp		0.49±0.11	(Conner and Day, 1991)
Bluebonnet swamp	42	0.41	(Sanders, 1998)
Heron Pond swamp	30	0.8	(Warren, 2001)
Pointe au Chene swamp	231	0.4	(Rybczyk et al., 1998)
Buttonland swamp	1600	0.25	(Demissie and Fitzpatrick, 1992)
La Union swamp	10	0.052	(Urquhart, 1999)
Tuckean Swamp	5000	0.22	(Taffs and Heijnis, 2008)
Nariva Swamp	6234[1]	0.31	(Ramcharan, 2004)
Loboi Swamp	150	0.1	(Ashley et al., 2004)

§ Area taken from US National wetland inventory

[1] http://www.ema.co.tt/new/images/guides/AppendixB.pdf

Appendix C

Supplementary Material: Chapter 4

Unit values from different datasets

The regional values of different biomes are taken from four studies (Table AC1) in the established regional dataset and finally the mean value is used for estimation of the value of ecosystem services.

The values of three ecosystem services (carbon sequestration, water purification and nutrient cycling) are extracted from the regional dataset (Table AC2). Some of the studies did not assign any value to these ecosystem services and some few assigned zero values. If there is only one value available for an ecosystem service that is referred as the best estimate. We also gathered 827 unit values from ESVD database (https://www.es-partnership.org/), standardized (to CAD 2017) and sorted regionally. Further, the mean unit value for different biomes is calculated. However, there is high variability in the minimum and maximum unit values which is shown by calculating the coefficient of variation (CV). Eventually, these values (Table AC3) are used to show the variation in the unit values from region to region.

Table AC 1: Unit values of biomes extracted from regional studies.

Biome	Regional unit values (2017 CAD/ha/year)				
	Greenbelt[1]	Southern Ontario[2]	Lake Simcoe[3]	Peace River[4]	**Mean**
Forest	6,660	5,150	5,900	1,440	**4,790±2315**
Grass/rangeland	1,990	410	3,350	1,150	**1,725±1260**
Wetlands	17,400	17,600	17,830	2,610	**13,760±7500**
Open water	410	5,860	1,800	230	**2,075±2620**
Cropland	590	340	1,170	1,275	**845±450**
Bare agriculture lands	2,050	-	-	-	**2,050**
Urban	-	-	1,015	-	**1,015**

[1](Wilson, 2008b); [2](Troy and Bagstad, 2010); [3](Wilson, 2008a); [4](Wilson, 2014)

Table AC 2: Unit values of ecosystem services for major biomes taken from regional studies.

Ecosystem Services	Providing Land Covers	Value ($/hectare/year)		
		Low	Best	High
Carbon Sequestration	Forest		48[a]	815[b]
	Pasture		35[a]	
	Wetlands		16[a]	
	Open water		16[a]	
	Cropland	0[a]	-	
	Bare agriculture lands		36[a]	
Water Purification	Forest	258[c]	583[a]	
	Pasture	0[c]		
	Wetlands	258[c]	583[a]	1701[b]
	Open Water	258[c]		
	Cropland	0[c]		
	Bare agriculture lands	0[c]		
Nutrient cycling	Forest	0[c]		596[e]
	Pasture/sparse forest	29[c]		28.5[e]
	Wetlands	0[c]	275[d]	3225[e]
	Open water	0[c]		63[e]
	Cropland	0[c]		-
	Bare agriculture lands			30[a]

[a](Wilson, 2008b);[b](Olewiler, 2004); [c](Wilson, 2008a); [d](Wilson, 2014); [e](Troy and Bagstad, 2010)

Table AC 3: Unit values for terrestrial biomes retrieved from ESVD database for various continents.

	Regional unit values of biomes (CAD 2017/ha/year)		
Continent/Biomes	Mean	SD	Coefficient of variation
Africa			
Cultivated	784	933	1
Forests [Temperate and Boreal]	308	376	1
Fresh water	1444	644	0.5
Grasslands	1	1	1
Americas			
Cultivated	239	131	1
Forests [Temperate and Boreal]	879	2328	3
Fresh water	1396	1883	1
Grasslands	123	168	1
Inland Wetlands	229639	460767	2
Tropical Forest	816	1165	1
Asia			
Forests [Temperate and Boreal]	477	0	0
Inland Wetlands	796	1151	1
Tropical Forest	522	1025	2
Europe			
Cultivated	1179	1791	2
Forests [Temperate and Boreal]	819	1839	2
Fresh water	2279	993	0.5
Grasslands	482	1144	2
Inland Wetlands	6886	11240	2
Latin America			
Cultivated	1592	2243	1
Forests [Temperate and Boreal]	164	302	2
Grasslands	1552	2581	2
Inland Wetlands	786	1357	2
Tropical Forest	518	1006	2
Oceania			
Cultivated	2	0	0
Inland Wetlands	1441	2244	2
Tropical Forest	1872	0	0
World			
Forests [Temperate and Boreal]	214	237	1
Fresh water	4746	7340	2
Grasslands	283	319	1
Inland Wetlands	7255	13145	2
Tropical Forest	1278	2194	2

I conducted the following t-Tests to assess the significant differences between the two mean values based on low and high resolution data:

Table AC 4: t-Test: Two-Sample Assuming Unequal Variances for forest

	Variable 1	*Variable 2*
Mean	94	84
Variance	20	15.16667
Observations	15	13
Hypothesized Mean Difference	0	
df	26	
t Stat	6.32455532	
P(T<=t) one-tail	5.35775E-07	
t Critical one-tail	1.705617901	
P(T<=t) two-tail	1.07155E-06	
t Critical two-tail	2.055529418	

As P <0.05, there is a significant difference between the two means.

Table AC 5: t-Test: Two-Sample Assuming Unequal Variances for Agriculture.

	Variable 1	*Variable 2*
Mean	57	56
Variance	256.6667	238.5
Observations	55	53
Hypothesized Mean Difference	0	
df	106	
t Stat	0.330289	
P(T<=t) one-tail	0.370917	
t Critical one-tail	1.659356	
P(T<=t) two-tail	0.741833	
t Critical two-tail	1.982597	

As P is not less than 0.05, there is not a significant difference between the two means.

Table AC 6: t-Test: Two-Sample Assuming Unequal Variances for total value of forest and agriculture

	Variable 1	*Variable 2*
Mean	151	140
Variance	275.5	256.6667
Observations	57	55
Hypothesized Mean Difference	0	
df	110	
t Stat	3.568871	
P(T<=t) one-tail	0.000266	
t Critical one-tail	1.658824	
P(T<=t) two-tail	0.000533	
t Critical two-tail	1.981765	

As $P<0.05$, there is a significant difference between the two mean values.

Appendix D
Supplementary Material: Chapter 5

We created a map of unit value distribution using the unit value of six major land use categories in the region (Figure A1). The extraction area is assigned no unit value (or 0 $/ha/year) because it does not generate ecosystem services.

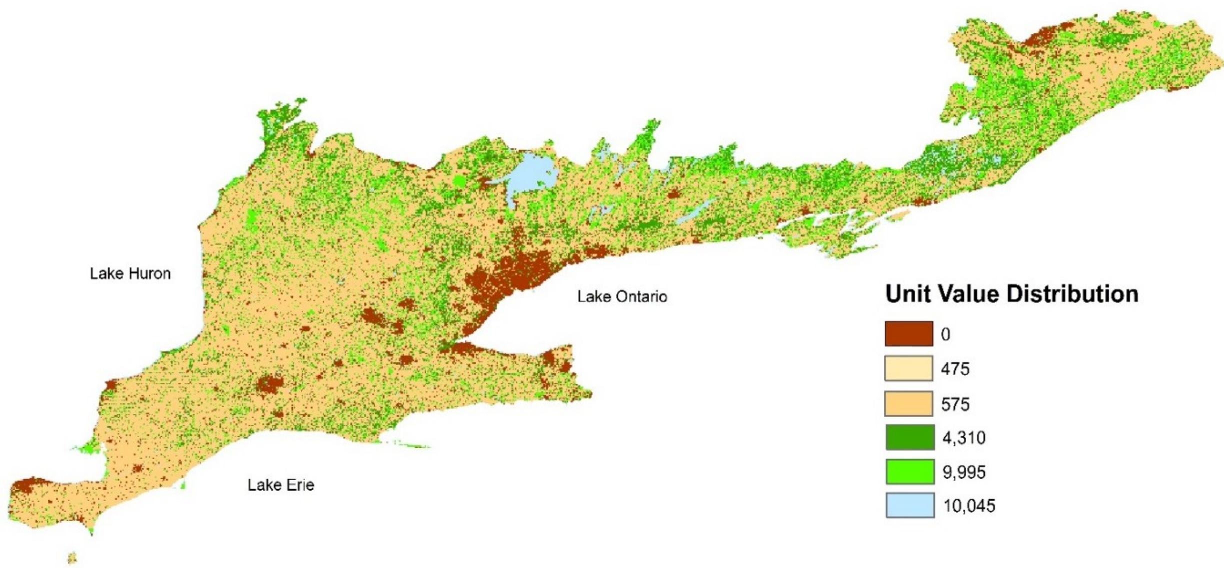

Figure AD 1: Unit value distribution map for a bundle of six ecosystem services in southern Ontario.