A Multi-Scale Modelling Approach to Valuing Ecosystem Services In and Around Long Point Biosphere Region

by

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Authors 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.

Abstract

Ecosystem services are the benefits that humans derive from ecosystems, and they deliver significant economic value to local and global beneficiaries, yet they are often underrepresented in conventional economic decision-making. Valuing ecosystem services makes the contributions of nature visible in decision-making arenas, which supports holistic evaluation of tradeoffs in the policy-making process. The objective of this study is to value diverse ecosystem services at three scales around Long Point Biosphere Region, which is a protected area along the north shore of Lake Erie and part of the World Network of Biosphere Reserves. To address this objective, this study employs replacement cost and benefit transfer methods to conduct these valuations.

The results of this study show, first, that the value of stormwater retention services provided by natural assets in Port Dover, a township near Long Point Biosphere Region, was found to be at least 2.6 million 2025 CAD per year. Second, the value of nutrient retention services provided by natural assets in the Long Point Region watershed was found to be at least 2.7 million 2025 CAD per year. Third, the value of a bundle of ecosystem services in the Long Point Biosphere Region Core was found to be 8 million to 87 million 2025 CAD per year. Each model was evaluated for resemblance to likely real-world ecosystem conditions. This analysis found that models for Port Dover and the Biosphere Core were defensible, while the model for the larger watershed was too different from likely real-world ecosystem conditions. These models are important to build narratives about the economic value of local ecosystems to community members, but cannot be used empirically in the absence of formal model validation.

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Introduction

Ecosystem services are the benefits that individuals retain from ecosystems (Mooney et al., 2005). It is important to value ecosystem services because it makes their value visible in conventional economic-decision-making (Costanza, 2024). Today, natural resources are explicitly excluded from Canadian public sector financial statements (Eyquem, 2025). This makes it difficult for governments to value ecosystem services and use these valuations to inform policymaking. Communities across Canada receive support from local organizations and academics to value ecosystem services in the absence of a formal financial mechanism for valuation.

This research values ecosystem services in Long Point Biosphere Region (LPBR) and Norfolk County. I use a multi-scale modelling approach to highlight the different scales at which ecosystem services are realized across the region and the diverse beneficiaries of these services.

First, I used replacement cost to value stormwater retention by natural assets in Port Dover, a community in Norfolk County with high impervious cover, as well as phosphorus retention by natural assets in the Long Point Region watershed. Replacement cost is a method used to value ecosystem services that considers the cost humans would incur to perform the same function as an ecosystem (Obst, n.d.; Selivanov & Hlaváčková, 2021). To model stormwater retention and phosphorus retention, I used the Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST®) tool (Natural Capital Project, 2025a). Next, I used benefit transfer to define the economic value of a bundle of ecosystem services

provided by the forested and watershed ecosystems in the LPBR Core. Benefit transfer applies a pre-existing value for a particular ecosystem or ecosystem service to a site of interest (Brander et al., 2013).

Valuing ecosystem services, especially using methods that capture exchange value, is important because it makes the value of environmental public goods visible in conventional economic decision-making. Standards like the System of Environmental Economic Accounting, which is a global standard for ecosystem accounting, provide guidance to policy-makers undertaking this work and enable fair comparisons between jurisdictions (United Nations, 2024a). These standards often require ecosystems to be valued using methods that represent exchange value.

This research was co-designed with LPBR to bring attention to the economic value that ecosystems provide to the community. In Norfolk County, local ecosystems provide services like erosion prevention, flood mitigation, and nutrient retention, but as public goods, the value of these services often goes under- or un-acknowledged. While valuing ecosystem services is generally useful to a municipality, there are two local circumstances that make the valuation of ecosystem services particularly relevant for LPBR and Norfolk County:

1. **Local infrastructure gaps (Antonacci, 2024)**- Norfolk County is currently experiencing gaps in stormwater infrastructure. Natural assets like wetlands and forests have been shown to effectively manage stormwater (Prudencio & Null, 2018) at a low-cost relative to conventional infrastructure. This valuation will draw

attention to the stormwater retention services that ecosystems provide to Norfolk County and point to low-cost alternatives like nature-based solutions to address these infrastructure gaps.

2. **Expanding LPBR's conservation paradigm (Environment and Climate Change Canada, 2023)-** Canada has committed to conserving 30% of land by 2030, as per the Kunming-Montreal Global Biodiversity Framework (KMGBF) (Government of Canada, 2024). Under this framework, areas that are not formally protected but are managed in a manner that promote the conservation of biodiversity can be categorized as Other Effective area-based Conservation Measures (OECM) and count towards KMGBF commitments (Environment and Climate Change Canada, 2022). There is interest in codifying other lands in the LPBR area of cooperation, including privately held land, as OECMs. Valuing ecosystem services will be an important ingredient for building the business case for conservation activities and compelling public and private stakeholders to participate in the OECM program.

This research takes a translational ecology approach (Enquist et al., 2017) by linking community needs to research direction. Translational ecology is a research approach that links natural sciences research to decision-making through collaboration with community stakeholders (Enquist et al., 2017). This research question was developed by community members around LPBR and research methods and scope were developed in consultation with staff at Long Point World Biosphere Reserve Foundation. Knowledge mobilization is a central component of this research: findings will be communicated to LPBR staff, local government, and community members via diverse deliverables, including reports and

presentations. These findings should support local decision-making and work to close the ecosystem services 'implementation gap' (Laurans et al., 2013; Levrel et al., 2017). Post-normal science ("Post-Normal Science," 2003) also informs this research, particularly in the pragmatic (Rea & Munns, 2017) and dialectic (Rizvi et al., 2024) lenses used to interpret the many values of nature. Post-normal science is a research discipline introduced by the ecological economics literature that investigates research problems with high systems uncertainty and high policy relevance ("Post-Normal Science," 2003).

Below, the findings of this research are discussed. First, ecosystem services literature and other relevant fields will be summarized. Next, the materials and methods used in this research will be documented. Then, results will be presented. Lastly, in the Discussion, implications, limitations, and future direction for this work will be highlighted.

Literature Review

What are Ecosystem Services?

According to the Millennium Ecosystem Assessment, ecosystem services are “the benefits people obtain from ecosystems” (Mooney et al., 2005). These benefits can typically be characterized according to one of four typologies (US Department of Agriculture, 2025):

- Provisioning services: goods or energy produced by an ecosystem, like water, food, or critical minerals
- Regulating services: the moderation and management of ecosystem processes, like erosion prevention or pollination
- Cultural services: immaterial benefits provided by ecosystems like existence value or recreational and spiritual benefits
- Supporting services: ecosystem processes that provide a foundation for the other three services, like biodiversity or water cycling

Most ecosystem services are public goods, which means that they are non-rival and non-exclusive (Simpson, 2016). Under our current market paradigm, public goods like ecosystem services remain both under-valued and under-provisioned because they are typically not bought and sold on markets and individual agents have limited incentive to contribute to their preservation (Kaul, 2012; Naidoo & Ricketts, 2006; Simpson, 2016). As a result of this perverse incentive structure, there has been a decline in global environmental integrity and rapid encroachment upon planetary boundaries (Richardson et al., 2023). To

move beyond an economic paradigm where the natural world exists merely as a market externality, ecosystem services can be valued to (more) accurately evaluate trade-offs between economic and ecological priorities (Costanza, 2024).

Below, I explore key areas of the literature that inform this research. First, the history of ecosystem services is discussed, followed by a summary of different methods used to value ecosystem services. Next, the implementation of ecosystem services and the many values of nature will be highlighted. Then, the ecosystem services landscape in Canada will be discussed, alongside a summary of the research site. Finally, the two epistemological foundations of this research, post-normal science and translational ecology, will be situated.

The History of Ecosystem Services

The limitations of traditional economic decision-making tools for evaluating tradeoffs between economic and ecological interests gained traction nearly 50 years ago, with the publication of a short essay in *Nature* that inquired about the value of nature's services. In this article, Dr. Walter Westman (1977) identifies the inter-dependency between human well-being and ecosystem function. He also calls attention to the disproportionate focus on valuing environmental stocks (or goods) rather than environmental flows (or services) in environmental economics. The ideas introduced in this essay continue to underpin ecosystem services literature today.

The next major contribution to this literature moved Westman's ideas from theory to practice when Costanza et al. (1997) conducted an economic valuation of global

ecosystem services. The authors estimate that the global value of just 17 ecosystem services to be 16-54 trillion USD/year, with an average value of 33 trillion USD/year (almost twice the value of gross national product at the time of publication) (Costanza et al., 1997). The authors updated this estimate to 125 trillion USD/year in 2011, projecting a 20 trillion USD/year reduction in productivity as a result of habitat degradation (Costanza et al., 2014). Costanza et al.'s (1997) article has been cited nearly 40,000 times and is widely considered to be one of the most influential works in ecological economics (Costanza et al., 2016).

In 2005, valuation of ecosystem services moved beyond academia and into global governance practice with the publication of the Millennium Ecosystem Assessment (Mooney et al., 2005). The Millennium Ecosystem Assessment was a global study that evaluated the linkages between ecosystem health and human well-being to build momentum for global conservation efforts. Following the Millennium Ecosystem Assessment, the System of Environmental-Economic Accounting (SEEA) and The Economics of Ecosystems and Biodiversity (TEEB) were both established to provide resourcing and guidance for valuing ecosystem services across the globe (The Economics of Ecosystems and Biodiversity, n.d.; United Nations, 2024a). SEEA is a global ecosystem accounting standard that can be integrated in the System of National Accounts that advances harmonized evaluation of ecosystem extent and function (United Nations, 2024a). TEEB is an initiative of the United Nations Environment Program that explores the economic value of nature (The Economics of Ecosystems and Biodiversity, n.d.).

Interest in ecosystem services valuation began to show marked growth following the publication of the Millenium Ecosystem Assessment and has continued to grow in the last two decades (Kubiszewski et al., 2022). Interest in the monetary valuation of ecosystem services rose alongside general interest in the field. Between 2012-2021, the use of the search terms 'ecosystem service value', 'economic value', and 'valuation' for publications related to ecosystem services grew by 350%, relative to 1984-2011 (Kubiszewski et al., 2022). A global synthesis by Brander et al. (2024) found that there was a dramatic increase in research output related to ecosystem services valuation in the last 10 years.

Methods for Valuing Ecosystem Services

There are a variety of methods that can be used to represent the economic value of ecosystems. Before highlighting the different methods for valuing ecosystem services, it is necessary to highlight the different components that contribute to the total economic value of ecosystems. Generally, the total of economic value of a particular ecosystem considers two parts: use value and non-use value (Mendes, 2012).

Use values reflect direct use and indirect use values, as well as the option value (Mendes, 2012). Direct use value describes the tangible benefits that can be derived from an ecosystem, indirect use value describes the tangible benefits provided through ecosystem function, and option values describe the value an ecosystem has with the understanding that it might be used in the future (Mendes, 2012).

Conversely, non-use values describe the intangible value individuals derive from ecosystems, including bequest value and existence value, which describe the benefit

derived from the knowledge that an ecosystem continues to exists, either for future generations (bequest value) or for personal benefit (existence value) (Mendes, 2012).

Generally, there are two families of methods for valuing ecosystem services in monetary terms (which look at the use and non-use values above): consumer surplus methods and exchange value methods. The exchange value is the price at which a service is or could be provided (Caparrós et al., n.d.), while consumer surplus reflects the difference between what a consumer is willing to pay and the exchange rate for a good or service (Ita, 2025). Methods for valuing ecosystem services that ask individuals what they are willing to pay or willing to accept represent consumer surplus, while methods that look at the price at which ecosystem goods or services are sold at or what price they could be sold at reflect the exchange value.

Consumer surplus values and exchange values are used for different purposes. Exchange values are used in ecosystem accounting and for other activities that follow global standards like SEEA (United Nations, 2024a). Conversely, methods that describe consumer surplus have been used in the policy and legal context to understand how individuals or communities value ecosystems (Carson et al., 2003; National Science Foundation & Environmental Protection Agency, 1998). Perhaps the most well-known use of consumer surplus methods was the contingent valuation study that informed the settlement between the State of Alaska and Exxon Valdez following the 1989 spill (Carson, 2012). Following the spill, a contingent valuation study was used to reach a $3 billion settlement paid to the state of Alaska by Exxon Valdez (Carson, 2012).

As discussed above, ecosystems hold two types of economic value: use value, which reflects direct and indirect use of ecosystems, and non-use values, which reflects intangible value individuals derive from ecosystems (Mendes, 2012). Both exchange value and consumer surplus methods can be deployed to evaluate the use value of ecosystems, while non-use value can be established only using a narrow range of consumer surplus methods called stated preference methods (Adamowicz et al., 1994; Ali et al., 2020). Stated preference methods value willingness-to-pay by asking individuals how much they would be willing to pay for a particular good or service (Costanza et al., 2011). The other group of consumer surplus methods are called revealed preference methods, where researchers analyze choices that consumers make to establish how much individuals are willing to pay for a particular good or service (Costanza et al., 2011).

Consumer surplus and exchange value methods study a particular site of interest. In sites with data or funding constraints, practitioners can use the benefit transfer approach, which applies the valuation conducted at one site (the "study site") to another site of interest (the "policy site") (Brander, 2013). Benefit transfer is one of the most popular methods for valuing ecosystems globally because it requires limited data and resources (Aziz, 2018; Richardson et al., 2014). Either exchange value or consumer surplus values can be applied using the benefit transfer approach. The value of an ecosystem or a particular ecosystem service is contingent upon a variety of factors, including biophysical variables, like ecosystem productivity and connectivity, and socioeconomic factors, like the number of beneficiaries (Troy & Wilson, 2006; Vačkář et al., 2018), so demonstrating

correspondence between study sites is necessary to conduct an effective benefit transfer (Plummer, 2009).

Common methods for valuing ecosystem services are described in Table 1.

Table 1: Common methods for valuing ecosystem services (King & Mazzotta, n.d.; Legesse et al., 2022; Mavsar et al., 2013)

Method	**Description**	**Valuation Method**
Contingent Valuation Method	Identifying the value of an ecosystem by asking people to state their willingness to pay to access said ecosystem.	Consumer surplus – stated preference
Hedonic Pricing Method	Attributing the influence of ecosystems on market prices for goods and services, particularly real estate, to assess willingness to pay.	Consumer surplus – revealed preference
Travel Cost Method	Determining the financial and non-financial costs people are willing to incur to access an ecosystem.	Consumer surplus – revealed preference
Market Price Method	Identifying the value of goods and services provided by an ecosystem in real-world markets.	Exchange value

Production Function-based Method	Attributing the role of goods and services from an ecosystem to the economic productivity brought about by other goods and services.	Exchange value
Damage Cost Avoided Method	Assesses the value of the infrastructure or activities that would be lost in the event of the ecosystem disappearing.	Exchange value
Replacement Cost Method	Represents the value of the ecosystem according to the cost that would be incurred to replace it.	Exchange value
Benefit Transfer Method	Applying a valuation from another location to an ecosystem of interest.	Varies; based on study site

Operationalizing Ecosystem Services: The Many Values of "Nature" and the Implementation Gap

The objective of valuing ecosystem services is to make the value of environmental public goods visible in conventional economic decision-making (Costanza et al., 2024). In public policy, ecosystem services valuations are used at three different policy stages (Pascual et al., 2023):

- **Informing** policy-makers and the public about the invisible value of ecosystems

- **Deciding** between different policy options and evaluating tradeoffs
- **Designing** policy, permits, and fees

Despite the policy relevance of ecosystem services valuation, there is a considerable 'implementation gap' between theory and practice in ecosystem services literature (Laurans et al., 2013; Levrel et al., 2017). The problem is twofold. First, academics generally do not prioritize the policy-relevance of their valuations, even though this is the stated objective of ecosystem services valuation (Laurans et al., 2013; Sitas et al., 2013). This disconnect is evidenced by the inattention to the use of ecosystem services valuations by policymakers within the literature (Laurans et al., 2013). As ecosystem services valuation explicitly aim to enhance conservation outcomes (Costanza, 2024), this neglect of policy-relevance is significant. Second, real-world implementation challenges, like regulatory barriers, inconsistent reporting requirements, and lock-in, prevent the use of ecosystem services valuation, even when researchers do center policy relevance and community partnership in their valuation (Eyquem, 2025; Kerr et al., 2021; Sitas et al., 2013).

Beyond challenges with implementation, it is challenging to value ecosystem services because of the need to distinguish between the instrumental and intrinsic value of nature. The intrinsic value of nature describes the right nature has to exist and the fundamental role that a function biosphere plays in human well-being (Costanza, 2024; Mooney et al., 2005). The intrinsic value of nature cannot be valued in monetary terms. Conversely, the instrumental value of nature describes the contribution of nature to particular outcome or goal (Costanza, 2024; Mooney et al., 2005). The use and non-use values of nature can be

considered instrumental value (Costanza, 2024; Mendes, 2012; Mooney et al., 2005). These values are not mutually exclusive, which means they exist simultaneously and acknowledging one should not detract from the other (Costanza, 2024). Valuing ecosystem services considers the instrumental value of nature, not the intrinsic value (Costanza, 2024).

There is a well-established sentiment that valuing ecosystem services is both reductive and anthropocentric (Costanza, 2024; Farley et al., 2024; McCauley, 2006; Nijnik & Miller, 2017; Schröter et al., 2014). This perception is rooted in the belief that identifying the instrumental value of nature excludes or minimizes nature's intrinsic value (Costanza, 2024). Critics worry that valuing ecosystem services will contribute to the commodification of nature (McCauley, 2006) and further erase the intrinsic value of nature in public discourses. This idea first emerged in the literature shortly after the publication of the Millennium Ecosystem Assessment (McCauley, 2006) and remains a well-known critique of ecosystem services valuation today.

Concisely, some people wonder why nature cannot just be preserved for its own sake. There is no shortage of literature that seeks to answer this question (Game et al., 2013; Hardin, 1968; Ostrom, 1990) and while this literature draws on a variety of disciplines, many authors come to the same conclusion: governing the commons in a global world under a market paradigm makes preserving nature for its own sake rather difficult. This challenge is primarily attributable to the social and psychological mechanisms that impede effective management of public goods in large user groups (Ostrom, 1990). Some economists believe that "in a world of trade-offs, valuation is not a choice" (Costanza,

2024, p. 4), but rather a tool to ensure that the invisible value of public goods is represented on the balance sheet in economic decision-making.

The intrinsic value of nature and the instrumental value of nature are not mutually exclusive (Costanza, 2024). Thus, valuing ecosystem services makes the economic benefits derived from environmental public goods visible (Pascual et al., 2023) but does not inherently detract from nature's intrinsic value (Ye et al., 2020). Emphasizing the many values of nature that exist in tandem is consistent with dialectical philosophy, which acknowledges the unavoidable polarity of the human experience (Rizvi et al., 2024). Adopting a dialectical approach to interpret the pluralistic value of nature has been embraced by environmental pragmatists who use market tools to advance environmental protection (Rea & Munns, 2017). Environmental pragmatism is a branch of environmental thought that supports pluralism and acknowledges that valuation, even in cases that could be construed as anthropocentric, can lead to positive environmental outcomes (Rea & Munns, 2017). Considering ecosystem services valuation through the lenses of dialectical philosophy and environmental pragmatism provides a philosophical foundation for the utility of valuing ecosystem services.

Valuing Ecosystem Services in Canada

It is estimated that the annual value of ecosystem services in Canada is 3.6 trillion CAD (Rice et al., 2018). Home to one-tenth of the global forested land and 7% of the world's renewable freshwater resources (Environment and Climate Change Canada, 2018; Natural Resources Canada, 2025b), Canada is composed of diverse ecosystems that support the

well-being of Canadians today and will continue to do so for future generations, if they are managed and conserved appropriately. Despite Canada's rich ecological fabric, efforts to value ecosystem services in Canada trail behind other countries and regions, like Western Europe, the United States, Brazil, and China (Acharya et al., 2019; Brander et al., 2024; Olukolajo et al., 2023).

At the federal level, Canada does report physical ecosystem extent and physical ecosystem accounts for SEEA, however, as of 2024, monetary accounts are not reported (United Nations, 2024a). Across the three levels of government, municipalities are leading this work in the Canadian context, enabled by programs like the Natural Assets Initiative and the Green Municipal Fund (Federation of Canadian Municipalities, n.d.; Natural Assets Initiative, 2025). Unfortunately, structural barriers remain that prevent appropriate integration of these valuations into policy and asset management processes. The most notable barrier is the regulatory exclusion of natural assets in Canadian public sector financial statements (Eyquem, 2025). Excluding the value of ecosystems from public financial records erases government accountability for ecosystem management and creates structural barriers to considering the instrumental value of ecosystems in economic decision-making processes (Eyquem, 2025).

In peer-reviewed literature, interest in valuing ecosystem services in Canada has remained steady but limited since the 1990s, contrary to increasing global interest (Macaskill & Lloyd-Smith, 2022). Valuation methods that include consumer surplus dominate the Canadian literature, particularly stated preference valuation studies: 81% of studies in

Canada conducted during the 2010s used stated preference methods (Macaskill & Lloyd-Smith, 2022).The underutilization of exchange value methods in Canadian ecosystem services valuation literature points to a notable research gap because of the important contributions of these methods for ecosystem accounting for various levels of government (Eyquem, 2025; United Nations 2024a), This disconnect between research and practice in the valuation of ecosystem services reinforces the implementation gap for ecosystem services work in the Canadian context (Kerr et al., 2021).

InVEST is one of the most popular tools for modelling ecosystem services with applications for exchange value (Natural Capital Project, 2025a). As of today, it has been used in only five academic articles to model ecosystem services in the Canadian context (Guerry et al., 2012; Kamga et al., 2024; Kim et al., 2012; Rioux et al., 2019; Singh et al., 2020).

Long Point Biosphere Region and Global Biosphere Reserves

LPBR is one of Canada's nineteen Biosphere Regions part of the larger UNESCO global biosphere reserve network (Government of Canada, 2025). In Canada, these regions are called Biosphere Regions rather than biosphere reserves (Government of Canada, 2025) because of the colonial connotation of the word 'reserve'. Located on the north shore of Lake Erie, LPBR represents both a physical space conserved through formal channels, as well as a non-profit that promotes local research and monitoring (Long Point Biosphere Region, n.d.). LPBR's core protected area is composed of Long Point National Wildlife Area and Backus Woods, but most of Norfolk County is a part of the larger area of cooperation for the Biosphere Region (Long Point Biosphere Region, n.d.). LPBR was designated as a

biosphere reserve in 1996 because it is a critical stopover site for migratory birds travelling along the Atlantic Flyway (Canada Biosphere Regions Association, n.d.). Like other Biosphere Regions, LPBR is composed of three areas (Canadian Biosphere Regions Association, 2025):

- Biosphere Core: The central protected area
- Buffer Zone: Surrounds the core area; extensions of critical research sites
- Area of Cooperation: Larger community that interacts with the Biosphere Region

There is a considerable community of researchers working in LPBR. Research activities are primarily confined to the natural sciences (Burrell et al., 2016; Lei et al., 2020; Loder et al., 2023; Robichaud & Rooney, 2017; Schummer et al., 2012). The value of local ecosystem services remains unstudied at LPBR. As a protected area and significant hub for local biodiversity (Canada Biosphere Regions Association, n.d.), LPBR is an ideal site for ecosystem services research because there is good data availability and research-focused staff with place-based expertise. Additionally, valuing ecosystem services at LPBR will highlight the return on investment from conservation regimes (Armsworth et al., 2018) and expand the business case for environmental conservation (Accounting for Sustainability, n.d.) to inform local decision-making, like the expansion of local OECMs.

Epistemological Foundations: Translational Ecology and Post-Normal Science

Post-normal science (PNS) and translational ecology (TE) are two interrelated research approaches with deep ties to the principles that underpin ecosystem services research and valuation (Ainscough et al., 2018; Wainger & Mazzotta, 2011). TE links academic

research to real-world decision-making by embedding community members across the research process (Enquist et al., 2017), while PNS addresses research questions with real-world implications in high levels of systems uncertainty ("Post-Normal Science," 2003). Both approaches center the science-policy interface (Lawson et al., 2017; Waylen et al., 2023), in order to bridge scientific knowledge and policy-making (Environmental Information, 2019).

PNS is a research discipline that bridges epistemology and governance for issue-driven science ("Post-Normal Science," 2003). For PNS systems uncertainty and decision stakes are high, whereas applied science research problems generally have low systems uncertainty and decision stakes ("Post-Normal Science," 2003). TE is a process-oriented research approach, which is characterized by involvement of non-academic stakeholders throughout the research process (Enquist et al., 2017). Both approaches have a strong commitment to using research to affect real-world problems to enhance sustainability (Enquist et al., 2017; "Post-Normal Science," 2003).

Despite the considerable overlap between these research approaches, TE and PNS have not been formally linked in the literature. Post-normal science emerged from ecological economics ("Post-Normal Science," 2003), so it has more readily been linked to ecosystem services (Ainscough et al., 2018; "Post-Normal Science," 2003), but there is some evidence of linking ecosystem services research to a translational ecology approach in the literature as well (Margeson et al., 2025). As both research approaches explicitly center policy-driven research and stakeholder engagement (Ainscough et al., 2018; Enquist et al., 2017), TE and PNS research approaches may be leveraged to address the

‘implementation gap’ in current ecosystem services research (Laurans et al., 2013; Levrel et al., 2017).

Methods and Materials

I used a multi-scale modelling approach (Boumans et al., 2015) to value ecosystem services at different scales around LPBR and Norfolk County. The multi-scale modelling approach is used to model complex systems, often linking human-environment or environment-economic supersystems (Boumans et al., 2015). In this case, ecosystem services are modelled at different scales across social, economic, and environmental super-systems, however, findings are not linked to investigate interactive effects or system-wide behaviour.

Two methods were utilized to reflect the diversity of local ecosystem services and their beneficiaries. First, I used replacement cost to value stormwater retention services in Port Dover, a community within Norfolk County, as well as nutrient retention services in the Long Point Region watershed. Replacement cost is a method used to value ecosystem services that considers the cost humans would incur to perform the same function as an ecosystem (Obst, n.d.; Selivanov & Hlaváčková, 2021). Second, I used benefit transfer to value wetland and forested ecosystems within the LPBR Core. Benefit transfer applies a pre-existing value for a particular ecosystem or ecosystem service to a site of interest (Brander et al., 2013). The benefit transfer considers a bundle of ecosystem services, including climate regulation, habitat provisioning, nutrient cycling, waste treatment, disturbance regulation, water supply, water regulation, pollination, pest and disease control, gas regulation, and erosion prevention (Mulrooney & Jones, 2023).

The research area was pre-determined by research partners at LPBR and the research methods were co-developed with the support of staff at the Biosphere Region. Community engagement was a central piece of this work, so various stakeholders were engaged throughout the research project, like municipal staff and conservation practitioners. The three study sites were also selected with consideration to the needs of the community. First, stormwater retention in Port Dover was selected because of the stormwater infrastructure crisis in the municipality (Antonacci, 2024). Next, phosphorus retention in the Long Point Region watershed was selected because of local phosphorus loading (Environment and Climate Change Canada, 2023). Lastly, the bundle of ecosystem services in the Biosphere Core was selected to build the business case for conservation and support the expansion of the OECM paradigm (Environment and Climate Change Canada, 2022). This research adopted a translational ecology approach (Enquist et al., 2017) to engage stakeholders across the research process and produce policy-relevant valuations.

Below, the methods and materials used to run the stormwater retention and nutrient retention models will be discussed using Refsgaard et al.'s (2007) environmental modelling process as a framework for introducing each step of the work (Figure 1). This framework describes the major steps in environmental modelling, with an emphasis on uncertainty in the modelling process (Refsgaard et al., 2007). After, the benefit transfer valuation will be discussed, outlining the benefit transfer approach, data inputs, and calculations.

Replacement Cost

Step One: Model Study Plan

The first step of the environmental modelling process considers problem identification, research scope and plan, and uncertainty analysis. Below, I outline the ecosystem services of interest and the problem context that contributed to their selection. Next, I discuss the geographic scope of the research. Then, I describe InVEST, which was used for ecosystem services modelling, and replacement cost, which was used to value the model outputs. Lastly, I discuss the uncertainty associated with this modelling work.

The two ecosystem services of interest for this work are stormwater retention and nutrient retention. Stormwater retention describes the uptake of rainfall by a surface, thereby preventing it from entering surface water as runoff (Cossin, n.d.). Ecosystems generally have greater retention rates than built-up land uses, like highways and buildings, and can be leveraged to manage stormwater during rainfall events (Kõiv-Vainik et al., 2022), thereby reducing total and peak stormwater input for human-made stormwater infrastructure. Nutrient retention describes the storage of excess nutrients in an ecosystem, preventing them from entering surface water as runoff (Hooftman et al., 2025). Excessive nutrient loading contributes to eutrophication, which has negative economic and ecological consequences (Environment and Climate Change Canada, 2023).

Both services have been chosen because they are relevant to current challenges faced by LPBR and the surrounding community, as identified in partnership with the LPBR. Stormwater retention services were chosen because the municipality is currently facing a

stormwater infrastructure crisis. Existing stormwater assets in the municipality require maintenance amounting to millions of dollars, putting further strain on existing budgetary pressures (Antonacci, 2024). Nutrient retention was selected because nutrient loading is a considerable concern for municipalities around the Great Lakes, driven largely by major agricultural economies like Norfolk County (Environment and Climate Change Canada, 2023). This research will highlight the role that natural assets can play in addressing both of these complex socio-ecological challenges for the community.

This research takes a multi-scale modelling approach, which means that ecosystems services are modelled at various scales. Stormwater retention was modelled for Port Dover, a community within Norfolk County. Stormwater retention by natural assets has the greatest economic utility when it lessens runoff to built stormwater assets. As Norfolk County is a rural community with low impervious cover, stormwater assets are not uniformly implemented across the region (Norfolk County, 2022). Port Dover was selected as the area of interest for this model because it has high impervious cover and a high concentration of stormwater assets (Norfolk County, 2022), so there is a clear link between retention by natural assets and reduced burden on built stormwater assets. The study area for stormwater retention modelling is in Figure 2 below.



Figure 1: Port Dover; area of interest for stormwater retention (Norfolk County, 2025) Map image is the intellectual property of Esri and is used herein under license. Copyright © 2025 Esri and its licensors. All rights reserved.

The nutrient retention model considers a much larger study area, focusing on the entire Long Point Region watershed. This is because phosphorus is a non-point source pollutant (Ge et al., 2023), which means that it cannot be traced back to a single source and can travel through ecosystems. Any area that is exposed to phosphorus runoff or that may be connected to an ecosystem exposed to phosphorus runoff can offer nutrient retention services by absorbing the nutrient into the landscape and preventing runoff into waterways. The study area for nutrient retention is available in Figure 3.

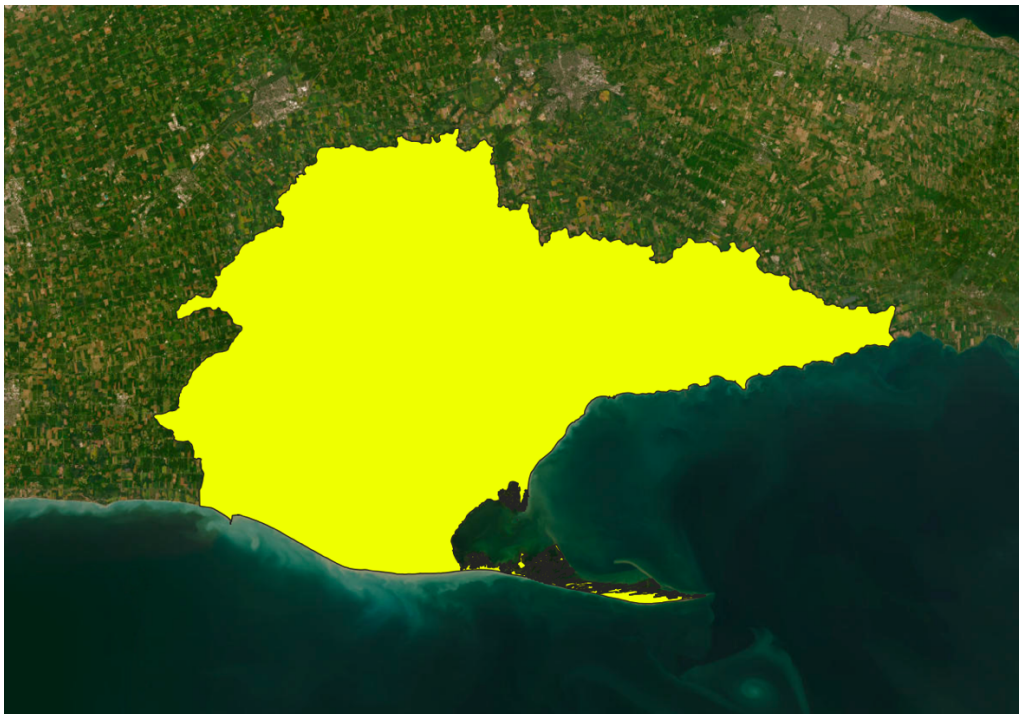


Figure 2: Long Point Region watershed, area of interest for nutrient delivery ratio (Government of Ontario, 2024) Map image is the intellectual property of Esri and is used herein under license. Copyright © 2025 Esri and its licensors. All rights reserved.

To conduct this valuation, stormwater retention and phosphorus retention services were modelled using InVEST, then results were monetized using the replacement cost approach. The InVEST tool and the replacement cost method are briefly described below.

The InVEST model (Natural Capital Lab, 2025) allows users to model 22 different ecosystem services using basic ecological and spatial data. As of May 2025, InVEST has been used in over 2,400 publications (Natural Capital Project, 2025b). In contrast to other tools used to model ecosystem services, InVEST has medium data demand and can be used by non-experts (Decsi et al., 2022), so it can be used by a wide variety of practitioners and in many spatial and temporal contexts. This versatility eliminates real-world barriers

for sustainability practitioners and scholars, making it a popular choice for ecosystem services modelling. Most datasets required to run InVEST can be found through the Natural Capital Project Data Hub, which is run by the Natural Capital Project to streamline the modelling process (Natural Capital Project, 2025c).

Replacement cost is the assumed value of a service provided by an ecosystem, if it were to be provided by a produced asset (Obst, n.d.; Selivanov & Hlaváčková, 2021). Values derived through replacement cost are considered appropriate for monetary ecosystem accounting because they represent exchange value and exclude consumer surplus (Edens & Graveland, 2024; United Nations, 2024b).

Replacement cost should utilize the least cost alternative to value an ecosystem service (López-Morales & Mesa-Jurado, 2017). The least cost alternative describes the lowest cost human-made intervention that can perform the same operation as the current ecosystem function. In this case, it is assumed that a single abatement measure would be applied with no constraints in the place of all natural assets. This is generally not possible in practice (Garcia-Hernandez et al., 2022), however, the replacement cost approach is applied here because it represents the most conservative estimate about the replacement cost of each ecosystem service.

Uncertainty is a key challenge for ecosystem services modelling. Ecosystem services are modelled and valued with the intention of influencing decision-making and policy outcomes (Laurans et al., 2013; Sitas et al., 2013), which underscores the need for accurate and defensible models. Insufficient model validation is a challenge for the

ecosystem services literature, mostly due to the complexity of ecosystem services, which makes monitoring expensive and, in many cases, infeasible (Pereira et al., 2025). In the absence of a corresponding monitoring regime, these models cannot be formally validated. Rather, each model will be evaluated to establish whether model results likely resemble real-world ecosystem conditions. This analysis will be informed by literature that explores stormwater retention and phosphorus retention services at similar study sites in the Discussion. Because the models cannot be formally validated, these results cannot be used empirically and should instead be used to build narratives about the value of ecosystem services in LPBR and Norfolk County.

Step Two: Data and Conceptualization

This section will discuss the data inputs as well as the model specifications. First, model specifications will be outlined. These will be linked to the geophysical, biophysical and economic data needs. Additionally, the sources for geospatial, biophysical and economic data will be discussed.

To estimate annual stormwater retention, InVEST's urban stormwater retention model uses a rainfall-runoff balance approach (Natural Capital Project, n.d.-b). This model calculates the volume of stormwater retention using the following equation:

$$V_{RE_i} = 0.001 \cdot P_i \cdot RE_i \cdot pixel.area \quad (1)$$

where V_{REi} is retention volume, P_i is annual precipitation and RE_i is the retention coefficient (Natural Capital Project, n.d.-b).

The same process is used to calculate runoff, using the following equation:

$$V_{RU,i} = 0.001 \cdot P_i \cdot RU_i \cdot pixel.\,area \tag{2}$$

where V_{RUi} is runoff volume, P_i is annual precipitation and RU_i is the runoff coefficient (Natural Capital Project, n.d.-b).

To estimate annual nutrient retention, InVEST's nutrient retention model uses a mass balance approach. First, they identify the modified phosphorus load per pixel, using the following equation:

$$modified.\,load_{x_i} = load_{x_i} \cdot RPI_{x_i} \tag{3}$$

where $loadx_i$ is the base phosphorus load and $RPIx_i$ is local runoff potential relative to the average value for the area of interest (Natural Capital Project, n.d.-a).

Next, the nutrient delivery ratio (NDR) is calculated, which represents the nutrient transport of each cell, using the following equation:

$$NDR_i = NDR_{0,i}\left(1 + \exp\left(\frac{IC_0 - IC_i}{k}\right)\right)^{-1} \tag{4}$$

where k and IC_0 are calibration parameters (Table 9) and IC_i is elevation relative to neighbouring pixels (Natural Capital Project, n.d.-a). InVEST does not calculate subsurface runoff for phosphorus, so only surface runoff is calculated.

Nutrient export is then calculated, using the following equation:

$$x_{exp_i} = load_{surf,i} \cdot NDR_{surf,i} + load_{subs,i} \cdot NDR_{subs,i} \quad (5)$$

where the product of surface load and NDR for each cell are calculated. Subsurface load and NDR are set to zero as the model cannot calculate subsurface values for phosphorus.

The stormwater retention model output is then valued, using the following equation:

$$TV_{SW} = V_{SW} * P_{SW} \quad (6)$$

where TV_{sw} is the total value of stormwater retention services, V_{sw} is the total volume of stormwater retained through natural assets in m^3 and P_{sw} is the cost per m^3 of stormwater management for grey infrastructure.

The nutrient retention model output is also valued, using the following equation:

$$TV_{PR} = V_{PR} * P_{PR} \quad (7)$$

where TV_{PR} is the total value of phosphorus retention services, V_{PR} is the total mass of phosphorus retained through natural assets in kg and P_{PR} is the cost per kg of nutrient retention through the least cost structural abatement measure.

To run both the stormwater retention and nutrient retention model, InVEST links geospatial data to biophysical information. It then aggregates the results into a shapefile. The data needs for the models are outlined in Table 2 and 3 below.

Table 2: Data needs for stormwater retention model (Natural Capital Project, n.d.-b)

Input	**Description**
Land Use/Land Cover (LULC)	Raster file that details land use for each pixel within the area of interest.
Soil Hydrologic Group	Raster file that details the soil group types within the area of interest, corresponding with the conventional soil hydrological grouping.
Precipitation	Raster file with average annual precipitation in mm.
Biophysical Tables	CSV file mapping land use types to runoff coefficients.
Area of Interest	Vector file that indicates the watershed of interest to aggregate model results.

Table 3: Data needs for nutrient delivery ratio model (Natural Capital Project, n.d.-a)

Input	**Description**
Land Use/Land Cover (LULC)	Raster file that details land use for each pixel within the area of interest.
Digital Elevation Model	Raster file of elevation.

Nutrient Runoff Proxy	Raster file that captures runoff potential; average annual precipitation appropriate.
Biophysical Tables	CSV file mapping land use to nutrient loading, retention efficiency, and critical length.
Area of Interest	Vector file that indicates the watershed of interest to aggregate model results.

Geospatial data inputs for this model were retained from databases managed by the Government of Ontario or other public bodies responsible for environmental data management. Biophysical data was in some cases available locally but often drawn from standards or resources from the United States. Geospatial and biophysical inputs for the stormwater retention model are described in Tables 4 and 5, and inputs for the nutrient retention model are described in Tables 6 and 7. Complete biophysical tables used to run each model are available in Appendix A and Appendix B.

Table 4: Geospatial data inputs for stormwater retention model

Input	**Source**
Land Use/Land Cover (LULC)	SOLRIS 3.0 (Government of Ontario, 2023)
Soil Hydrologic Group	Soil Survey Complex (Ontario Ministry of Agriculture, Food and Agribusiness, 2023)
Precipitation (mm/year)	Mean annual precipitation (ClimateNA, n.d.)

Area of Interest	Conservation authority administrative areas (Government of Ontario, 2024)

Table 5: Biophysical table inputs for stormwater retention model

Input	**Description**	**Source**
Connected Impervious Surface (T/F)	0 = false, 1 = true LULCs 201, 202, and 203 = 1, else = 0	Manual recommendations
Stormwater coefficient by soil group	Please see Appendix A for the value assigned to each LULC.	(Alaska Department of Environmental Conservation Division of Water, 2024; CivilWeb, n.d.)

Table 6: Geospatial inputs for nutrient delivery ratio model

Input	**Source**
Digital Elevation Model	Medium Resolution Digital Elevation Model of Canada (NRCan, 2025)
Land Use/Land Cover (LULC)	SOLRIS 3.0 (Government of Ontario, 2023)
Nutrient Runoff Proxy	Mean annual precipitation (ClimateNA, n.d.)

Watershed	Conservation authority administrative areas (Government of Ontario, 2024)

Table 7: Biophysical table inputs for nutrient delivery ratio model

Input	**Description**	**Source**
Load	Load set to 0 for all LULC except 193 (tilled), which was set to 0.59 kg/ha	Plach et al, 2019
Load Type (option)	Load type set to 'measured runoff', based on the methods outlined in Plach et al, 2019	Manual recommendations
Retention Efficiency	Ratio set to 0.8 for forested ecosystems, 0.6 for other natural ecosystems, 0.05 for built up areas and 0 for undefined areas and water bodies, as well as agricultural lands	InVEST v 3.9 manual recommendations (Sharp et al., 2020)
Critical Length	Set to 30.	Manual recommendations

Monetary values for ecosystem services were established through a review of the literature. Stormwater retention was valued with a proxy value of 1.59 USD (Simpson & McPherson, 2007 as cited in Natural Capital Project, n.d.-b) per m^3 of stormwater. Assuming the average 2007 exchange rate (1.07 CAD = 1 USD) and average inflationary effects (48% percent increase) (Bank of Canada, 2025; Canada Revenue Agency, 2017), this value is 2.53 CAD today. Phosphorus retention was valued using an analysis of phosphorus abatement practices in Ontario (Liu, 2017). The structural abatement measure with the lowest mean economic cost was selected, which had an abatement cost of 36.85 CAD per kg of phosphorous. Assuming average inflationary effects (26% increase) (Bank of Canada, 2025), this value is 46.43 CAD today.

Step Three: Model Set-up

This research does not develop a novel modelling tool, rather it utilizes an existing tool using local data.

To prepare spatial data for the models, all files were reprojected to NAD27 / UTM zone 17N and clipped according to the Long Point Region watershed boundaries. The NAD27 / UTM zone 17N projection was selected because it is based on a North American datum and suitable for storing data using a linear coordinate system, as required by the model (MapTiler, 2025).

Both the Soil Survey Complex and the mean annual precipitation files required additional data transformation. The Soil Survey Complex, which was used for the stormwater retention model, was initially stored as a vector file. In order to rasterize this file, the soil

hydrological group values, which were expressed using the conventional alphabetical groupings (USDA, 2016) were transformed to be represented numerically (Table 8), then the vector file was rasterized using the QGIS rasterize tool. The mean annual precipitation file, which was used for both models, was initially missing data towards the east of the study area for the stormwater retention model. To fill this gap, the raster file was reprojected according to the boundaries of the area of interest and the raster file was resampled to assign an average precipitation value to the entire raster file.

Table 8: Hydrological group rasterized values

Initial Soil Hydrological Group Value	**New Soil Hydrological Group Value**
A	1
B	2
C	3
D	4
N	4

Step Four: Calibration and Validation

Model calibration and validation are essential to ensure that environmental models accurately reflect real-world ecosystem conditions. Below, calibration will be discussed, followed by model validation.

InVEST differs from conventional ecosystem services models because it considers complex biophysical processes in a simplified way (Desci et al., 2014), so there are less calibration parameters in InVEST models compared to traditional ecosystem services modelling tools (El-Sadek & İrvem, 2014). The stormwater retention model does not have any calibration parameters (Natural Capital Project, n.d.-a). The nutrient retention model has three parameters: threshold flow accumulation, Borselli's K and flow direction algorithm. Each is defined in Table 9. The values selected for each value are outlined in Table 10.

Table 9: Calibration parameters for nutrient delivery ratio model (Natural Capital Project, n.d.-a)

Specification	**Description**
Threshold Flow Accumulation	Experimental value used to establish stream footprint. This value is visually compared to actual stream footprint to calibrate.
Borselli K	Describes "the shape of the relationship between hydrologic connectivity… and the nutrient delivery ratio" (Natural Capital Project, n.d.-a). This parameter is expressed numerically.
Flow direction algorithm	Two algorithms can be used to establish flow direction. The two algorithms are:

	D8: assumes that water flows to only one neighbouring pixel MFD: assumes that water flows to all downslope neighbours

Table 10: Model specifications settings for nutrient delivery ratio model (Natural Capital Project, n.d.-a)

Input	**Description**	**Source**
Threshold Flow Accumulation	Set to 1000.	Intermediate raster output compared to Ontario Hydro Network (Ontario Ministry of Natural Resources and Forestry, 2025)
Borselli K Parameter	Set to 2.	Manual recommendations
Flow Direction Algorithm	Set to D8.	Manual recommendations

Model validation is important because it demonstrates that real-world ecosystem processes are being represented accurately. Using independent field data to validate environmental models is considered the best approach (Refsgaard et al., 2007). As discussed in Step One, there is not an accompanying ecosystem services monitoring paradigm, so the models were not formally validated. Rather, correspondence of each

model to likely real-world ecosystem conditions is highlighted in the Discussion. An exploratory review of the literature was conducted, considering local studies, similar study sites, and meta-analyses, to establish likely real-world ecosystem conditions.

Step Five: Simulation and Evaluation

Below, the final and intermediate outputs for each model will be discussed.

The InVEST model results are presented in a shapefile with results for the area of interest. For stormwater retention, these values are (Natural Capital Project, n.d.-a):

- Mean retention ratio
- Total retention volume (m^3/year)
- Mean runoff ratio
- Total runoff volume (m^3/year)

For nutrient retention, these values are (Natural Capital Project, n.d.-b):

- Phosphorus surface load (kg/year)
- Phosphorus surface runoff (kg/year)

Each model also returns intermediate files and raster files that can be used to analyze spatial patterns in the model. For stormwater retention, these files include (Natural Capital Project, n.d.-a):

- **retention_ratio.tif**: stormwater retention ratio for each pixel
- **retention_volume.tif**: stormwater retention volume for each pixel
- **runoff_ratio.tif**: stormwater runoff ratio for each pixel
- **runoff_volume.tif**: stormwater runoff volume for each pixel

- **lulc_aligned.tif**: LULC file aligned with soil group and precipitation files
- **soil_group_aligned.tif**: soil group file aligned with LULC and precipitation files
- **precipitation_aligned.tif**: precipitation file aligned with soil group and LULC files

For nutrient retention, these files include (Natural Capital Project, n.d.-b):

- **p_surface_export.tif**: final export rate for each pixel (kg/ha/yr)
- **load_p.tif**: total phosphorus load for each pixel (kg/ha/yr)
- **stream.tif**: stream network created by the model, for model calibration (see Tables 8-9)

Benefit Transfer

Benefit transfer is one of the most common methods used to value ecosystem services because it requires limited primary data input (Richardson et al., 2014). Benefit transfer allows researchers to apply the value of an ecosystem found elsewhere to a site of interest (Brander et al., 2013). This method considers two locations:

- Study site: a location that was used to derive a value for an ecosystem service
- Policy site: the location of interest that offers the same ecosystem service

The value of the study site is applied to the policy site, usually on a per unit basis, either per capita or per unit of space. Equation 8 below describes the general approach:

$$TV_i = PSA_i \cdot SSA_i \quad (8)$$

where TV_i is total value ($), PSA_i is policy site value per unit ($/ha or $/person) and SSV_i is study site value (ha or people). Given the limited data needs, the results of benefit transfer are widely deployed to consider the value of ecosystems in economic decision-making activities like cost-benefit analysis (Johnson & Button, 1997).

The policy site for this research is the Long Point Region Biosphere Region Core (Figure 4). Specifically, the benefit transfer values forested and wetland ecosystems. The study site values are derived from a prior valuation of Canada's national parks (Mulrooney and Jones, 2023). These values were selected because they demonstrate correspondence to the policy site (Plummer, 2009) as they address Canadian protected areas. The study site values are available in Table 11 below. They have been adjusted to account for the rate of inflation from the initial year of publication (4% increase) (Bank of Canada, 2025).



Figure 3: Long Point Biosphere Region Biosphere Core (Long Point Biosphere Region, 2024) Map image is the intellectual property of Esri and is used herein under license. Copyright © 2025 Esri and its licensors. All rights reserved.

Table 11: Study site values for wetland and forested ecosystems per hectare (Mulrooney & Jones, 2023), adjusted to 2025 CAD

	Mulrooney & Jones (2023) Value ($/ha)		
	Low	**Med**	**High**
Wetland Ecosystem	$3,918	$35,606	$67,293
Forested Ecosystem	$4,739	$18,590	$32,441

To conduct the valuation, spatial extent of the wetland and forested ecosystem data (DMTI, 2024a; DMTI, 2024b) was clipped according to the Biosphere Core boundaries (Long Point Biosphere Region, 2024). The area of each ecosystem was then measured using the attribute table. The study site values were then multiplied by the total area, as per Equation 8.

Results

The models found that ecosystems provide considerable utility to LPBR, the larger area of cooperation, as well as the Long Point Region watershed. Below, I outline the findings of the replacement cost modelling conducted via InVEST, which values stormwater retention in Port Dover, as well as phosphorus retention for the broader Long Point Region watershed. The findings of the benefit transfer for the wetland and forested ecosystems in the Biosphere Core are also presented.

Replacement Cost

Stormwater Retention

This model found that Port Dover (4.3 km^2) receives about 4,093,935.125 m^3 of rainfall every year. A total of 1,031,830.625m^3 is retained by ecosystems, while 3,062,104.5 m^3 is not. This means that natural assets within Port Dover reduce the stormwater burden on municipal assets by approximately 25%. The approximate replacement cost for stormwater retention is 2.53 2025 CAD per m^3. Thus, natural assets in Port Dover provide 2,609,680.52 2025 CAD in stormwater retention services annually.

Natural land uses were responsible for the majority of stormwater retained, with a retention ratio of up to 95% compared to 4% retention for built up, impervious land use (Figure 5).

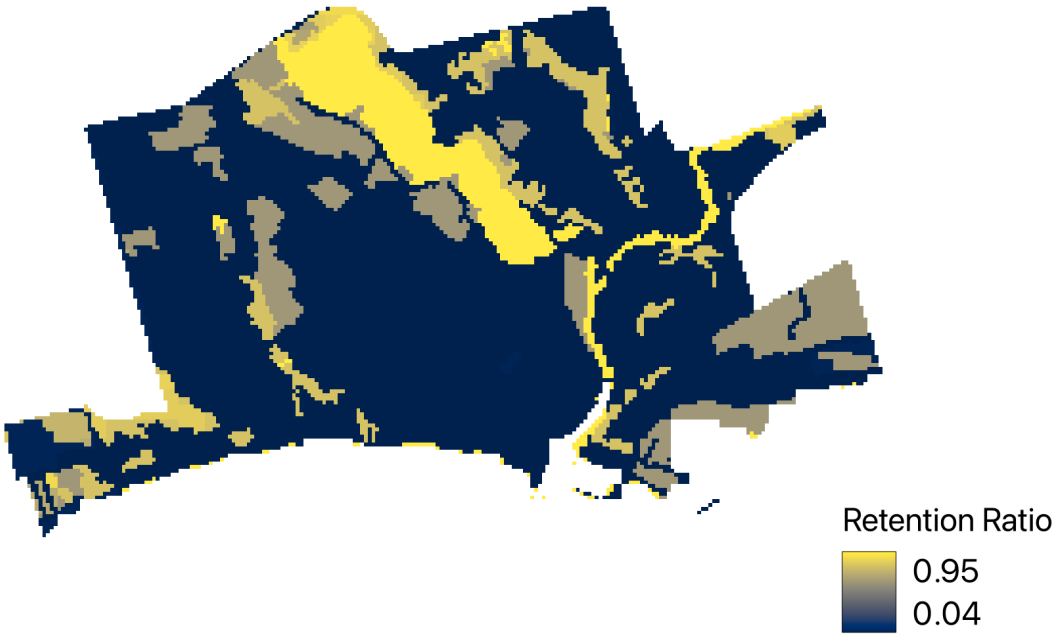


Figure 4: Retention ratio in Port Dover

To run the model, the various input files were aligned to perform operations. Aligned soil group and LULC rasters are available in Figures 6 and 7 below. Descriptions of each LULC code can be found in the appendices. A mean annual precipitation raster was also produced, but all cells were assigned a value of 966.81 mm per year.

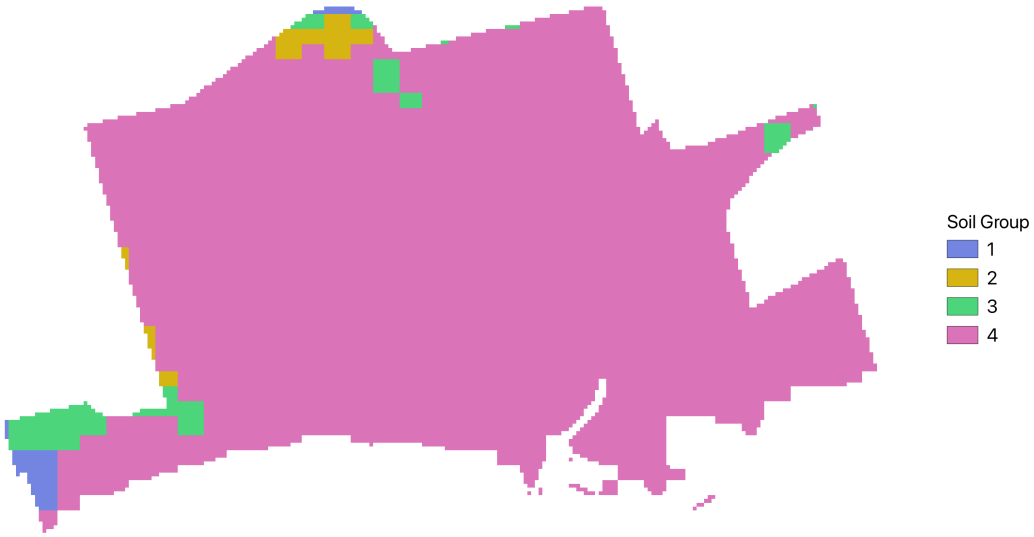


Figure 5: Aligned soil groups in Port Dover

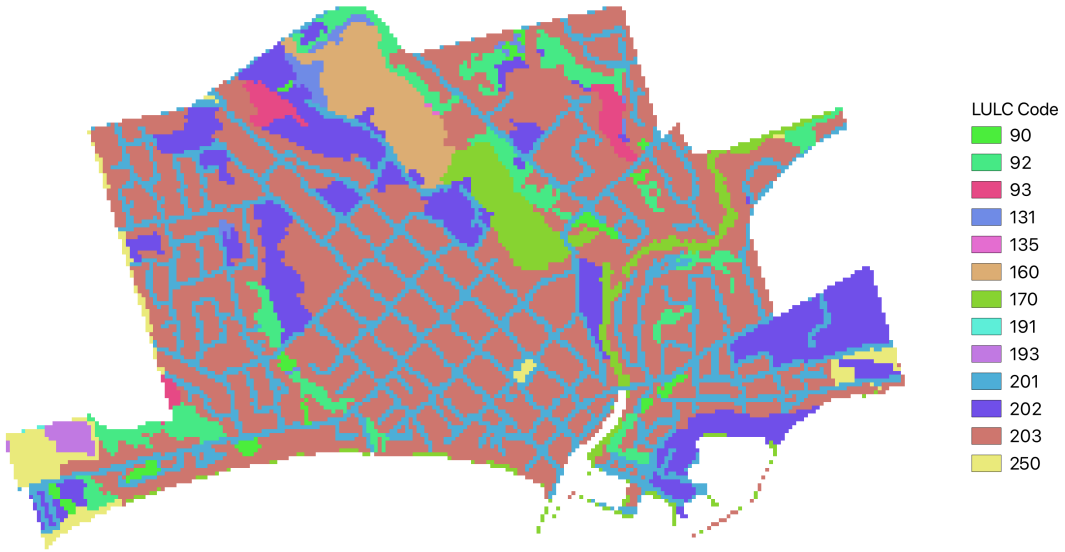


Figure 6: Aligned Land Use/Land Cover in Port Dover

Nutrient Retention

The model found that the total annual edge-of-field runoff across the Long Point Region watershed (2844 km^2) was 83,920 kg of phosphorus per year, while total annual export was 26,551 kg of phosphorous and total landscape retention was 57,368 kg of phosphorous. Edge-of-field runoff describes the nutrients that leave agricultural land uses as runoff (Plach et al., 2019). These findings suggest that 68% of phosphorus introduced to the ecosystem through agricultural runoff is absorbed into ecosystems and that about 32% is exported from the ecosystem. The approximate replacement cost for nutrient retention is 46.43 2025 CAD per kg of P. Thus, natural assets in the watershed provide 2,663,653.61 2025 CAD in nutrient retention services annually.

A minority of edge-of-field runoff from each cell eventually reaches a stream via surface flow (Figure 8).

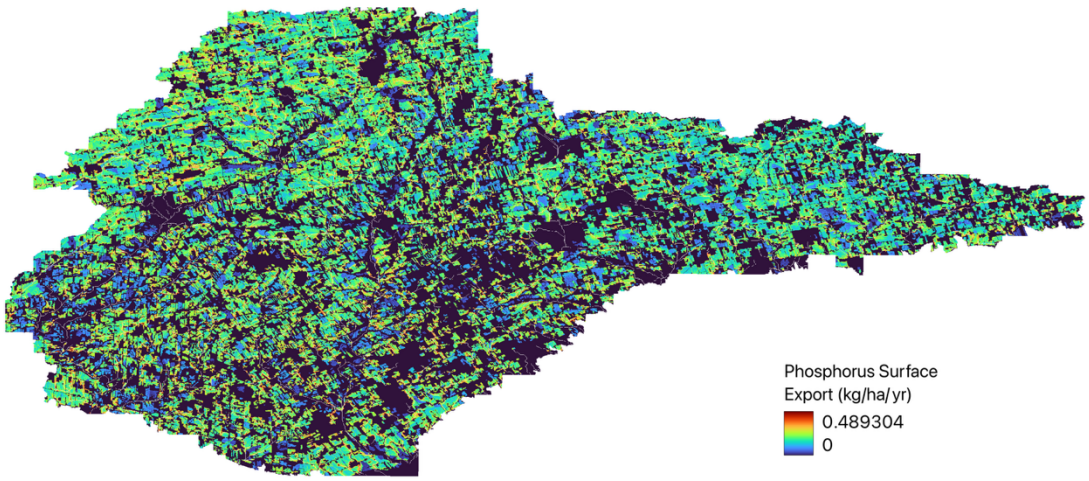


Figure 7: Phosphorus surface export from Long Point Region watershed

This model only evaluated retention of edge-of-field runoff, so the phosphorus load for all agricultural land classes was 0.59 kg/ha/yr and 0 kg/ha/yr for all other land classes (Figure 9).

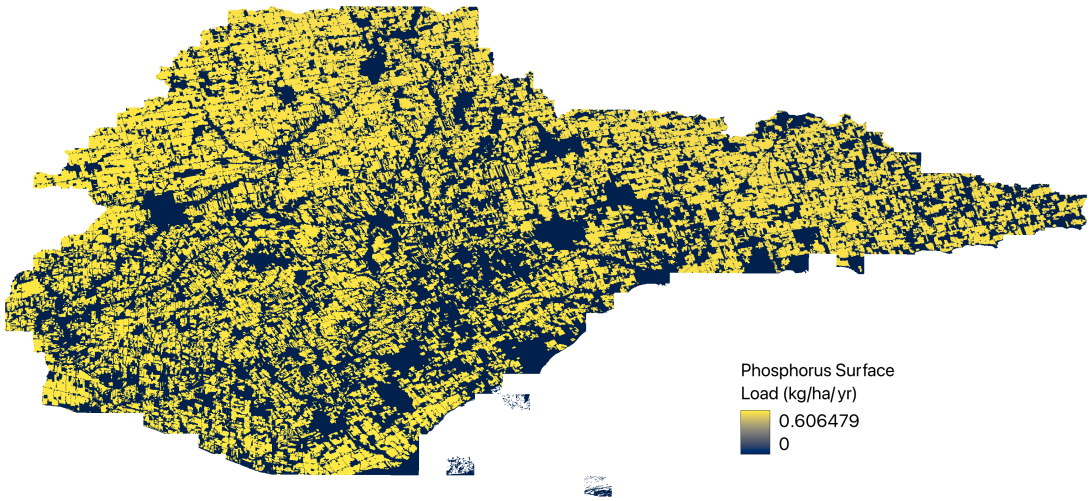


Figure 8: Phosphorus surface load for Long Point Region watershed

The threshold flow accumulation parameter was used to calibrate the model to the actual river footprint. The stream network in Figure 10 was produced as an intermediate output for this model, which sufficiently resembles the actual river footprint (Ontario Ministry of Natural Resources and Forestry, 2025), to accept the chosen threshold flow accumulation value.

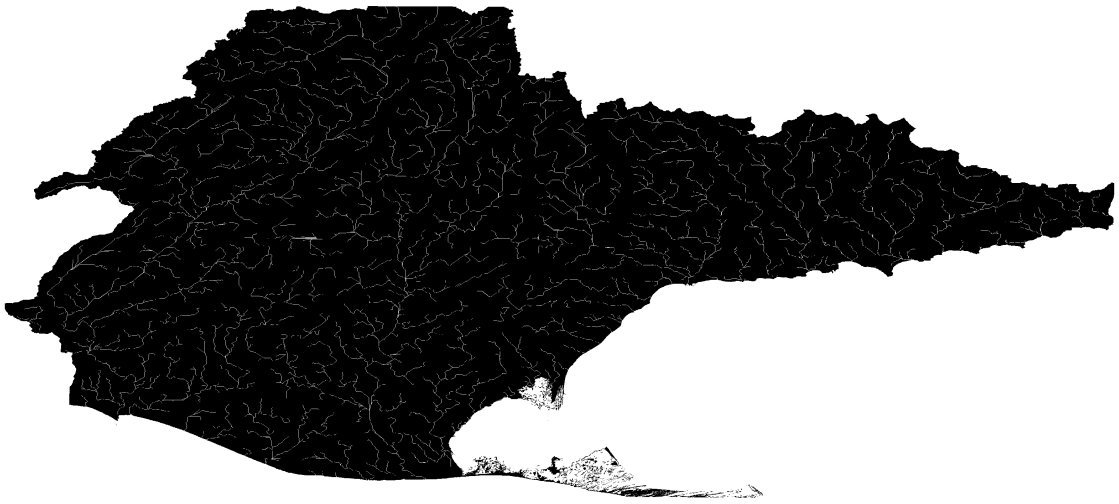


Figure 9: Intermediate stream network for Long Point Region watershed

Benefit Transfer

This analysis found that the Biosphere Core is home to about 1198.32 ha of forested ecosystems and 714.87 ha of wetland ecosystems. The annual economic value of these ecosystems, as per the values adopted by Mulrooney and Jones (2023), is 8-87 million 2025 CAD (Table 12). These values encompass a range of ecosystem services, including climate regulation, habitat provisioning, nutrient cycling, waste treatment, disturbance regulation, water supply, water regulation, pollination, pest and disease control, gas regulation and erosion prevention.

Table 12: Value of forested and wetland ecosystems in Long Point Biosphere Region Core, all values adjusted to 2025 CAD

	Low	**Med**	**High**
Wetland Ecosystem	$2,800,632	$25,454,004	$48,105,890
Forested Ecosystem	$5,679,174	$22,276,769	$38,874,364
Total Value	**$8,479,806**	**$47,730,773**	**$86,980,253**

Discussion

The results of this study demonstrate that ecosystems provide considerable economic value to stakeholders across the zone of cooperation for LPBR. Below, the general findings of the model will be restated and compared to likely real-world ecosystem conditions. Additionally, the correspondence between the study site and the policy site for the benefit transfer will be discussed. Following this critical analysis of the models, the implications of this work for Long Point Biosphere Region and Norfolk County, as well as the Canadian ecosystem services literature will be discussed. Next, the limitations of these models will be discussed. Lastly, a roadmap for integrating ecosystem services and other environmental public goods into conventional economic decision-making in Canada will be discussed, which was informed by this research.

The absence of model validation is a persistent problem in the ecosystem services modelling literature (Pereira et al., 2025; Willcock et al., 2023), even though validating environmental models against field data is a key step in the modelling process (Refsgaard et al., 2007). For InVEST models, validation rates in the peer-reviewed literature are as low as 13% (Valladares-Castellanos et al., 2024). When integrated into decision-making processes, it is essential that ecosystem services models and values are validated against field data to promote appropriate ecosystem management and build credibility for this emerging field (Pereira et al., 2025).

For this research, formal validation was not possible without a corresponding monitoring paradigm. Instead, data collected at other scales and other sites will be used to establish

whether the models are expected to reasonably represent likely real-world ecosystem conditions. As a result of the absence of site-level model validation, the valuations from this study cannot be used empirically (i.e., to make asset management decisions), but they can be used to construct narratives about the monetary value provided by local ecosystems for community members and local decision-makers.

The stormwater retention model found that about 25% of annual precipitation in Port Dover was retained by ecosystems. Exploration of the literature suggests that this model likely reflects real-world ecosystem conditions for public natural assets but underrepresents the value of privately-owned natural assets (Baker, 2022; Ongaga et al., 2024). The stormwater retention ratio (Figure 5) corresponds with LULC values (Figure 7), assigning very low retention rates to built-up impervious surfaces, moderate retention rates to built-up pervious surfaces, and high retention rates to aquatic ecosystems, like marsh and open water. This is consistent with general trends in the literature that suggest that areas with high impervious cover have much greater runoff rates than their impervious counterparts (Ongaga et al., 2024). Conversely, marshes and open water are both aquatic ecosystems that receive stormwater runoff, so high retention rates are expected. The main weakness of this model is the characterization of residential properties in the LULC input file. Within Port Dover, SOLRIS classifies all residential areas as impervious built-up areas (Figure 7). Most residential properties in Port Dover have a front and back yard, as observed via Google Maps (2025), which means that each lot contains a considerable area of permeable surface not accounted for in the model. This means that the model underestimates stormwater retention by privately held natural assets. Therefore, I

conclude that the model sufficiently represents likely real-world behaviour of public natural assets, however, it excludes most retention by privately owned natural assets. This limitation must be stated in any application of these results.

The nutrient retention model found that ecosystems in the Long Point Region watershed absorbed about 68% of edge-of-field phosphorus runoff. Exploration of the literature suggests that the retention rate may resemble likely real-world ecosystem conditions, but that total phosphorus loading is considerably lower than likely real-world conditions (Environment and Climate Change Canada, 2023; Van Staden et al., 2021). For example, a recent study found that about 87.5 ± 5% of annual net anthropogenic phosphorus input applied to agricultural land uses in Ontario watersheds is absorbed by the landscape (Van Staden et al., 2021). This study reported that the Long Point Region watershed has moderate phosphorus loading and runoff rates relative to other watersheds, suggesting that the average landscape retention value for Ontario adequately represents the area of interest (Van Staden et al., 2021). This study considers the total volume of absorbed phosphorus, including by agricultural land uses, while the InVEST model considers edge-of-field phosphorus runoff only (Van Staden et al., 2021). The landscape absorption reported in the Van Staden et al (2021) model is ~30% greater than the InVEST model results, which likely accounts for agricultural phosphorus absorption. As agricultural phosphorus absorption is not captured by the InVEST model, this considerable difference between the models could suggest that the phosphorus retention model may resemble likely real-world ecosystem conditions. However, mass-based phosphorus loading data suggests that this model is inaccurate. In 2022, it was estimated that 2,091,000 kg of

phosphorus entered Lake Erie from Canada (Environment and Climate Change Canada, 2023). Our model finds that 26,551 kg of phosphorus left the watershed every year as runoff, amounting to about 1.3% of total Lake Erie phosphorus loading. Long Point Region watershed is one of only 7 administrative watersheds along the Canadian shoreline of Lake Erie (Government of Ontario, 2024). It is also considerably larger than most of the other watersheds with Lake Erie shoreline, like Catfish Creek and Kettle Creek (Government of Ontario, 2024). This suggests that the model underrepresents true phosphorus loading in the Long Point region watershed. Given the magnitude of difference between the model findings and the likely phosphorus loading attributable to this watershed, this model cannot be assumed to reasonably represent likely real-world ecosystem conditions.

The benefit transfer found that the value of a bundle of ecosystem services in the Biosphere Core was 8-87 million 2025 CAD per year. Benefit transfer usually takes place when budget is limited or in regions with data scarcity (Plummer, 2009), so considering correspondence between the study site and the policy site is appropriate in lieu of model validation. In this study, the study site values were derived from a valuation of Canada's national parks (Mulrooney & Jones, 2023). The sites demonstrate regional correspondence, as they are both in Canada, as well as functional correspondence, as they are both protected areas. However, there are some limitations to the ecological correspondence of the sites. Mulrooney & Jones' (2023) valuation considers all forests to have a uniform value, however, there are 8 distinct forest regions across Canada (Natural Resources Canada, 2025a). LPBR is home to Carolinian forest, which is distinct from the Boreal forest that blankets most of Canada (Natural Resources Canada, 2025a). While the Carolinian

forest is primarily composed of deciduous trees, the Boreal forest is primarily composed of coniferous trees (Natural Resources Canada, 2025). Other studies have demonstrated that coniferous and deciduous trees differ in their ability to provide ecosystem services (Mashizi & Sharafatmandrad, 2023). In summary, the two habitats do demonstrate regional and functional correspondence, but there are limitations to the ecological correspondence of the forested ecosystems.

The above discussion demonstrates that the stormwater retention model is defensible, while phosphorus retention model is not. The stormwater retention model is defensible because it reasonably represents likely real-world ecosystem conditions for public natural assets, based on previous literature. Yet since the phosphorus retention values are much smaller than likely real-world ecosystem conditions, based on previous literature, this result may not be accurate. The benefit transfer model demonstrates mixed correspondence: they are regionally and functionally similar, but the forested ecosystems do differ ecologically. Despite these limitations, the values produced in this research are appropriate for building narratives about the value of ecosystem services (i.e., to illustrate the invisible value of ecosystems to decision-makers and community members) rather than for conducting empirical analysis (i.e., to inform any economic or asset management decisions).

Contribution to Canadian Ecosystem Services Modelling Literature

This research contributes to Canadian ecosystem services modelling literature by piloting the use of the InVEST stormwater retention and nutrient retention models in Ontario. As a

key tool for modelling the exchange value of ecosystem services, demonstrating that InVEST models can be utilized in the Canadian context with existing data should help close the implementation gap (Laurans et al., 2013; Levrel et al., 2017) and promote policy-relevant ecosystem services valuations. Below I will first highlight the literature gap, then I will discuss research implications, and finally, I will discuss areas for future research.

There is insufficient attention paid to exchange value methods in the Canadian literature. Most Canadian research uses consumer surplus methods (Macaskill & Lloyd Smith, 2022), which have limited utility for local and regional governments in the Canadian context because ecosystem accounting standards that guide public valuation of ecosystem services require exchange value methods (Eyquem, 2025; United Nations, 2024a). This means that most valuations in the literature will not be used by governments to enhance economic outcomes for ecosystems and natural resources. Similarly, InVEST has been underutilized in Canada: a review of the literature identified only five academic articles that used InVEST to model ecosystem services in the Canadian context (Guerry et al., 2012; Kamga et al., 2024; Kim et al., 2012; Rioux et al., 2019; Singh et al., 2020). None of these articles focus on Ontario. The overrepresentation of consumer surplus methods and the underutilization of InVEST are both a product of the insufficient attention paid to exchange value methods in the Canadian literature. Because exchange values are the most useful to governments, these literature gaps reinforce the broader implementation gap in the Canadian ecosystem services literature.

This research addresses this implementation gap by demonstrating applications of exchange value methods (replacement cost) and tools (InVEST) in the Canadian context.

This is the first study that uses InVEST as a tool for modelling ecosystem services in Ontario and is also one of the first studies that uses InVEST in the Canadian context. InVEST can be used to model the exchange value of ecosystem services, so demonstrating the application of InVEST models in the Canadian context points to additional tools that can be used to value ecosystem services using diverse methods. This should help address the implementation gap in Canadian ecosystem services literature (Laurans et al., 2013; Levrel et al., 2017).

While this research begins to address this implementation gap through the use of the InVEST model, future research on valuing ecosystem services are needed to close the implementation gap in Canadian ecosystem services literature. Therefore, I recommend two areas for future research. First, future studies of Canadian ecosystem services should focus on the exchange value of natural assets to ensure that research is policy relevant. Using InVEST is one tool that can be used to model the exchange value for ecosystem services and close the implementation gap. Second, to reinforce the relevance of InVEST in the Canadian context, future researchers should also consider deploying InVEST models alongside a monitoring paradigm to formally validate future ecosystem services models.

Community Implications

This research adopted a multi-scale modelling approach to illustrate the different scales at which ecosystem services are realized and to highlight the different beneficiaries that realize these benefits. Below, the potential benefits of each model's result will be briefly discussed.

In Port Dover, ecosystems provide stormwater retention services. The primary beneficiaries of this ecosystem service are the municipal government and landowners, who have a reduced stormwater infrastructure burden and reduced flood risk (Zoghi et al., 2025). This is especially pertinent given Norfolk County's stormwater infrastructure crisis (Antonacci, 2024). This reduction in stormwater runoff translates to reduced municipal spending on grey stormwater infrastructure, which benefits the municipality and taxpayers, as well as reduced property damage and insurance claims due to stormwater detention in natural assets reducing peak runoff, which benefits homeowners, small businesses, and families.

Across the Long Point Region watershed, ecosystems also provide phosphorus retention services. The model assumes that farmers are the primary beneficiaries of these savings, because the replacement cost considers farm-level structural abatement measures. However, phosphorus loading and eutrophication have broad macroeconomic impacts across the Great Lakes Region (Garcia-Hernandez et al., 2022), so a range of beneficiaries realize material benefits from the phosphorus retention services provided by ecosystems, including industries, communities, and families around Lake Erie. These groups realize savings from avoided human health burden, reduced water treatment costs, and stable property values because of reduced phosphorus loading and eutrophication (Dodds et al., 2008).

Lastly, the Biosphere Core provides a range of ecosystem services, including climate regulation, habitat provisioning, nutrient cycling, waste treatment, disturbance regulation, water supply, water regulation, pollination, pest and disease control, gas regulation and

erosion prevention (Mulrooney & Jones, 2023), which provide utility to local and global beneficiaries. Climate regulation generally benefits global beneficiaries by reducing total climate change-related costs (Bilal & Känzig, 2024). Most other ecosystem services provided by the Biosphere Core, like habitat provisioning, nutrient cycling and pollination, are primarily associated with local benefits, like the provisioning of crops, which brings incomes to farmers (Khalifa et al., 2021), and tourism income (Grandview Research, 2025), but have global implications for planetary boundaries overshoot for freshwater change, land system change, biosphere integrity, and biogeochemical flows (Richardson et al., 2023).

Limitations of this Work

Despite the contribution that this research provides to the valuation of ecosystem services in practice, there are several limitations of the study. The first limitation is the absence of a corresponding monitoring paradigm. The second limitation is related to the data quality and availability for the benefit transfer. The final limitation is the inaccuracy of the phosphorus retention model.

The first limitation of this work is the absence of a corresponding monitoring paradigm. To validate environmental models, real-world ecosystem data should be used (Refsgaard et al., 2007). Without a corresponding monitoring paradigm, it is not possible to validate the stormwater retention model or the nutrient retention model. Lack of model validate is a persistent challenge in the ecosystem services modelling literature (Pereira et al., 2025;

Willcock et al., 2023). For InVEST models, validation rates in the literature are as low as 13% (Valladares-Castellanos et al., 2024).

The second limitation of this work is the relevance of the benefit transfer. There are two primary limitations of the benefit transfer. First, as discussed above, Mulrooney and Jones (2023) consider Canada's forested ecosystems as a monolith. As the LPBR is a primarily deciduous forest, while most of Canada's forests are coniferous, there may be different ecosystem function by forested ecosystems at the policy site relative to the study site. Additionally, most of the benefit transfer values used by Mulrooney and Jones (2023) were adopted from the grey literature. This introduces data quality concerns, as well as data accessibility concerns. Some of the resources used by the authors were taken offline, which makes it difficult to verify or evaluate data quality.

The final limitation of this work is the inaccuracy of the phosphorus retention model. In the Results section, I determine that the model considerably underrepresents likely phosphorus loading. I believe that this is primarily a result of inaccurate edge-of-field runoff values. The edge-of-field runoff were derived from study sites outside of the Long Point Region watershed (Plach et al., 2019), therefore the runoff values function only as a proxy for true edge-of-field runoff. Additionally, the study investigates just a handful of properties that utilized certain best management practices (Plach et al., 2019), which means that the findings may not be generalizable to a larger community of farmers.

Despite these limitations, the findings of this research are still useful for LPBR and Norfolk County, as well as the academic community. The values can be used to construct

narratives about the value of ecosystems for residents and decision-makers, rather than for empirical analysis. Even with these limitations, this research still points towards tangible utility provided by ecosystem services that will help support better policy and build a business case for OECMs. This research pilots the use of InVEST's stormwater and phosphorus retention models in Ontario, signalling to the academic community that this is a useful tool for valuing ecosystem services via replacement cost, which should help address the implementation gap (Laurans et al., 2013; Levrel et al., 2017) in Ontario's ecosystem services literature.

Policy Recommendations

Despite the foundational role ecosystems play in the promotion of human well-being, ecosystem services are often invisible in traditional economic decision-making processes (Costanza, 2024; Mooney et al., 2005). To make the benefits provided by natural assets visible, governments must make regulatory changes to recognize the value of ecosystem services, embed these values in economic and policy arenas, and harmonize valuation approaches across jurisdictions to allow for fair comparisons between different market contexts. Below, I explore how ecosystem services are currently considered in economic decision-making and present two policy recommendations that would unlock improved integration of environmental public goods to enhance policy outcomes.

In Canada, the value of public environmental goods is often not expressed in monetary terms for policy or financial processes. At the federal level, Canada is a party to SEEA, so high level ecosystem accounting is reported on an annual basis (United Nations, 2024a),

but there are structural barriers that prevent more holistic integration of the economic value of ecosystem services into decision-making processes. Today, natural resources are explicitly excluded from Canadian public sector financial statements (Eyquem, 2025), which is a considerable barrier to integrating the monetary value of ecosystems into public decision-making. Modifying these regulations to include natural assets would unlock a tangible mechanism to reflect environmental public goods on the balance sheets for governments across Canada. Once included in balance sheets for municipalities and upper levels of government, the value of natural assets could be leveraged across the policy process, including by informing policy-makers of the invisible value of ecosystems, monetizing ecological tradeoffs during decision-making processes, and enhancing policy and permitting design (Pascual et al., 2023). Currently, some municipalities include the monetary value of natural assets in unaudited financial statements (Eyquem, 2025), but these efforts are fragmented and inconsistent. By modifying the public sector financial statement regulatory framework, municipalities and other levels of government would be compelled, or even required, to include the monetary value of natural assets in audited financial statements.

In addition to unlocking mechanisms to recognize the value of nature through regulatory change, standards or tools are needed to ensure that jurisdictions are valuing environmental public goods using a common approach. Today, a fragmented approach is adopted in different jurisdictions across Canada as different tools and methods are used to value the same ecosystem services (Eyquem, 2025). This duplicates efforts and costs, prevents comparisons across jurisdictions, and opens the door for bad faith actors to

underrepresent the true economic value of natural assets. Interim standards for integrating ecosystem services into unaudited municipal financial reports have been developed (Eyquem, 2025), but the Standards Council of Canada would need to build on these standards to ensure that a harmonized approach is adopted across jurisdictions.

In summary, to reflect the invisible value of ecosystem services in economic decision-making processes, the regulatory framework for public sector financial statements must be modernized and a formal tool or standard must be developed to streamline valuation across jurisdictions. Modifying the regulations for public sector financial statements would allow the value of ecosystems to be expressed monetarily, enabling fair comparisons between natural assets and other economic goods. Introducing a standard or tool would enable a consistent and accurate approach for valuation across jurisdictions to reduce expenses and streamline valuation processes. These two policy recommendations could transform the Canadian economic paradigm to include environmental public goods. Improving management of local ecosystems would enhance the utility of natural assets, promote conservation, and promote cost savings for families and governments across Canada.

Conclusions

Ecosystem services are the benefits that humans realized from ecosystems (Mooney et al., 2005). As public goods, the contributions of ecosystem services are often underrepresented in our existing economic paradigm. Valuing ecosystem services is a proven strategy for making the value of ecosystem services visible (Costanza, 2024). This study found that ecosystems in and around LPBR provide considerable economic value to multiple beneficiaries. The stormwater retention model found that ecosystems in Port Dover provide at least 2.6 million 2025 CAD in stormwater retention services per year, the phosphorus retention model found that ecosystems in Long Point Region watershed provide at least 2.7 million 2025 CAD in phosphorus retention services per year, and the benefit transfer found that a collection of ecosystem services in the Biosphere Core provide 8-87 million 2025 CAD in benefits every year. While the model for phosphorus retention did not sufficiently correspond with likely real-world ecosystem conditions, the stormwater retention model sufficiently reflects expected ecosystem function. The benefit transfer sites do demonstrate functional and regional correspondence but do have some level of ecological differences. Overall, the stormwater retention model and the benefit transfer model are both acceptable representations of the economic value of ecosystems in and around LPBR.

This study contributes to local knowledge, as well as the literature. First, this study responds to a community need for the valuation of ecosystem services. This research has valued ecosystem services at three scales in and around LPBR for community members to

construct narratives about the economic contributions local ecosystems make to human well-being. This should enhance local decision-making and allow local conservation actors to make the business case for OECMs to compel private landowners to participate in the program. In addition to community contributions, this research also makes contributions to the academic literature. In the Canadian context, exchange value methods are underrepresented in the ecosystem services valuation literature (Macaskill & Lloyd-Smith, 2022). Consequently, the InVEST model, which focuses on exchange value, has also been underutilized in the Canadian context. This study makes a meaningful contribution in advancing exchange value methods in the Canadian context and pioneers the use of InVEST modelling in Ontario. Further research should continue to expand the use of exchange value methods to address the implementation gap in Canadian ecosystem services literature (Laurans et al., 2013; Levrel et al., 2017).

Today, municipalities and other levels of government cannot consider the true value of ecosystems in conventional economic decision-making because ecosystems are excluded from public sector financial statements (Eyquem, 2025). In the Canadian context, these regulations must be modified, and a standard should be introduced to enable governments to consider the true value of environmental public goods in economic and policy arenas.

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Appendix A: Stormwater Retention Biophysical Table

lucode	luname	is_connected	rc_a	rc_b	rc_c	rc_d
11	Open Beach/Bar	0	0.55	0.55	0.55	0.55
21	Open Sand Dune	0	0.55	0.55	0.55	0.55
23	Treed Sand Dune	0	0.55	0.55	0.55	0.55
41	Open Cliff and Talus	0	0.94	0.94	0.94	0.94
43	Treed Cliff and Talus	0	0.94	0.94	0.94	0.94
51	Open Alvar	0	0.94	0.94	0.94	0.94
52	Shrub Alvar	0	0.94	0.94	0.94	0.94
53	Treed Alvar	0	0.94	0.94	0.94	0.94
64	Open Bedrock	0	0.94	0.94	0.94	0.94
65	Sparse Treed	0	0.25	0.34	0.42	0.5

81	Open Tallgrass Prairie	0	0.25	0.34	0.42	0.5
82	Tallgrass Savannah	0	0.25	0.34	0.42	0.5
83	Tallgrass Woodland	0	0.25	0.34	0.42	0.5
90	Forest	0	0.11	0.14	0.16	0.2
91	Coniferous Forest	0	0.11	0.14	0.16	0.2
92	Mixed Forest	0	0.11	0.14	0.16	0.2
93	Deciduous Forest	0	0.11	0.14	0.16	0.2
131	Treed Swamp	0	0.05	0.05	0.05	0.05
135	Thicket Swamp	0	0.05	0.05	0.05	0.05
140	Fen	0	0.05	0.05	0.05	0.05
150	Bog	0	0.05	0.05	0.05	0.05
160	Marsh	0	0.05	0.05	0.05	0.05

170	Open Water	0	0.05	0.05	0.05	0.05
191	Plantations – Tree Cultivated	0	0.18	0.21	0.25	0.29
192	Hedge Rows	0	0.18	0.21	0.25	0.29
193	Tilled	0	0.18	0.21	0.25	0.29
201	Transportation	1	0.96	0.96	0.96	0.96
202	Built-Up Area-Pervious	1	0.29	0.32	0.35	0.38
203	Built-Up Area-Impervious	1	0.96	0.96	0.96	0.96
204	Extraction-Aggregate	0	0.94	0.94	0.94	0.94
205	Extraction–Peat/Topsoil	0	0.05	0.05	0.05	0.05
250	Undifferentiated	0	0.94	0.94	0.94	0.94

Appendix B: Nutrient Delivery Ratio Biophysical Table

lucode	luname	load_p	load_type_p	eff_p	crit_len_p
11	Open Beach/Bar	0	measured-runoff	0.05	30
21	Open Sand Dune	0	measured-runoff	0.05	30
23	Treed Sand Dune	0	measured-runoff	0.05	30
41	Open Cliff and Talus	0	measured-runoff	0.05	30
43	Treed Cliff and Talus	0	measured-runoff	0.05	30
51	Open Alvar	0	measured-runoff	0.05	30
52	Shrub Alvar	0	measured-runoff	0.05	30
53	Treed Alvar	0	measured-runoff	0.05	30
64	Open Bedrock	0	measured-runoff	0	30
65	Sparse Treed	0	measured-runoff	0.6	30
81	Open Tallgrass Prairie	0	measured-runoff	0.6	30
82	Tallgrass Savannah	0	measured-runoff	0.6	30

83	Tallgrass Woodland	0	measured-runoff	0.6	30
90	Forest	0	measured-runoff	0.6	30
91	Coniferous Forest	0	measured-runoff	0.6	30
92	Mixed Forest	0	measured-runoff	0.6	30
93	Deciduous Forest	0	measured-runoff	0.6	30
131	Treed Swamp	0	measured-runoff	0.8	30
135	Thicket Swamp	0	measured-runoff	0.8	30
140	Fen	0	measured-runoff	0.8	30
150	Bog	0	measured-runoff	0.8	30
160	Marsh	0	measured-runoff	0.8	30
170	Open Water	0	measured-runoff	0	30
191	Plantations – Tree Cultivated	0	measured-runoff	0.5	30
192	Hedge Rows	0	measured-runoff	0.5	30
193	Tilled	0.59	measured-runoff	0	30
201	Transportation	0	measured-runoff	0.05	30
202	Built-Up Area-Pervious	0	measured-runoff	0.05	30

203	Built-Up Area-Impervious	0	measured-runoff	0.05	30
204	Extraction-Aggregate	0	measured-runoff	0.05	30
205	Extraction–Peat/Topsoil	0	measured-runoff	0.8	30
250	Undifferentiated	0	measured-runoff	0	30