A Theoretical and Empirical Investigation into Payments for Watershed Ecosystem Services

by

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Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners. I understand that my thesis may be made electronically available to the public.

Statement of Contributions

Chapter 2 is co-authored with my co-supervisor, Professor Roy Brouwer. For this project, I contributed to the conceptualisation, data collection, empirical analysis, and writing, while Professor Brouwer provided support with the conceptualisation, writing, analysis, and guidance throughout.

Chapter 3 is co-authored with my co-supervisors, Professors Roy Brouwer and Alain-Désiré Nimubona. I contributed to the additional data collection (Professor Brouwer had already requested the original data from Statistics Canada), empirical analysis, and writing. Professor Brouwer provided support with the conceptualisation, empirical analysis and writing throughout. Professor Nimubona provided feedback on the draft paper. This paper was accepted and presented by me at the 29^{th} Annual Conference of the European Association of Environmental and Resource Economists in Leuven, Belgium in July, 2024.

Chapter 4 is co-authored with my co-supervisors, Professors Alain-Désiré Nimubona and Roy Brouwer. The original idea for the project was developed by myself, and I contributed to the data collection, analysis, and writing. Professor Nimubona provided support to ensure the mathematical notation was correct and the writing throughout. Professor Brouwer provided feedback on the draft paper. A draft version of this paper was presented at the Annual Conference of Canadian Resource and Environmental Economics Association in Saskatoon in September, 2022.

Abstract

This thesis contains three research chapters on the economics of Payments for Watershed Services (PWS). While each chapter covers a different aspect of PWS schemes, all three provide insights into the management of uncertainty in ecosystem services. The first chapter serves as an introduction to the problem and the main research question addressed by each paper.

Forested watersheds provide a variety of ecosystem services. Their economic valuation has increased significantly over the past decades, but the literature is fragmented and heterogenous and little has been done to systematically analyse estimated values. This study presents a global analysis of the economic valuation of forested watershed services. We address two methodological issues in the literature: the impact of ecosystem service classification on value estimates and sensitivity to scale. The latter is measured as the forested watershed area size compared to common practices to measure overall area size including other land cover and use. In the former case, we compare the detailed Common International Classification of Ecosystem Services with more simple and informal classifications in the literature based for example on the Millenium Ecosystem Assessment. We show that both the explanatory and predictive power of the estimated meta-regression models increase as we include more details about the valued ecosystem services and use more accurate estimations of the forested area size. Findings are, where possible, cross-validated with the existing forest hydrology literature. The study highlights the economic significance of maintaining forest cover in watershed areas and emphasises the role economics can play in determining high-value uses for ecosystem services.

The second chapter utilises the 2015 Survey of Drinking Water Plants, employing spatial re-

gression models and mediation analysis to examine the relationships between land use, raw water quality, and treatment costs. The study reveals a significant and economically substantial impact of forest cover on reducing treatment costs, primarily through its influence on turbidity levels. Forest cover significantly reduces turbidity levels, thereby decreasing treatment costs. The results indicate that converting agricultural land to forest within a 5km buffer zone around a treatment facility can generate savings of $18.92 CAD per hectare per year, while the savings relative to urban cover can reach up to $21.37 CAD per hectare per year. These savings diminish as the distance from the facility increases, with lower per hectare savings observed at the 10km buffer and sub-sub drainage basin scales. Further, the study accounts for spatial auto-correlation and the effects of wildfires on treatment costs. Wildfires are shown to lead to substantial increases in turbidity, significantly impacting treatment costs. The spatial error and lag models used highlight the importance of considering spatial dependencies in ecosystem service valuations. This research underscores the economic value of forest conservation and management in supporting water treatment processes and provides valuable insights for policymakers and stakeholders in water resources management. The study represents a significant step forward in understanding the interplay between land use, water quality, and treatment costs in Canada.

The main objective of the final study presented here is to develop a novel modelling framework in the context of Payments for Ecosystem Services (PES) to mimic and simulate behaviour of agents (e.g., landowners) providing ecosystem services and a principal (e.g., government, municipality) buying them under uncertainty. Uncertainty is defined as the case where the principal and agents lack precise knowledge on the parameters that govern ES output, but knows the range these parameters belong to. We compare contracts under two different decision-making paradigms, namely standard and robust optimisation. With robust optimisation, an uncertainty-averse principal designs a contract to maximise their worst-case outcome and obtain a performance guarantee, i.e., a minimum acceptable performance. The results show that with standard, input-based contracts, the only way for a principal to achieve this guarantee is to invest conservatively in ES. In this setting, the result holds with adverse selection and moral hazard. However, when input is

unobservable, using output-based payments, the principal can achieve this performance guarantee only by sharing some of the value of ES output with the landowner.

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Table of Contents

Examining Committee Membership ii

Author's Declaration iii

Statement of Contributions iv

Abstract v

Acknowledgements viii

1 Introduction 1

1.1 Forested Watersheds and the Provision of Drinking Water 8

2 Testing the Effect of Ecosystem Service and Land Classification on Global Values of Forested Watershed Ecosystem Services 15

2.1 Introduction 15

2.2 Literature Review 18

2.3 Econometric model, data collection and database 23

2.3.1 Econometric model 23

2.3.2 Data collection 24

2.3.3 Database 25

2.4 Results 30

2.4.1 Summary Statistics 30

2.4.2 Estimated meta-regression models 32

2.5 Conclusions . . . 36
2.6 Appendix . . . 38
2.6.1 List of studies . . . 38
2.6.2 PRISMA 2020 Flow Diagram of Data Collection Process . . . 39

3 Estimating The Value of land use Choices on Drinking Water Treatment Costs: A Spatial Regression Approach 40

3.1 Introduction . . . 40
3.2 Literature Review . . . 43
3.3 Methodology . . . 46
3.4 Data . . . 50
3.5 Results . . . 55
3.5.1 Spatial Regression Models . . . 55
3.5.2 Structural Equation Models . . . 59
3.6 Conclusion . . . 63
3.7 Appendix . . . 66
3.7.1 Adjustment of standard errors by effective sample size . . . 66
3.7.2 Spatial Effects . . . 67

4 Contracting Under Uncertainty in Payments for Ecosystem Services 69

4.1 Introduction . . . 69
4.2 Literature Review . . . 73
4.3 Expected Utility Analysis . . . 76
4.3.1 First-Best Allocation . . . 76
4.3.2 Adverse Selection . . . 79
4.4 Robust Contract . . . 80
4.4.1 First-Best Allocation . . . 80
4.4.2 Adverse Selection . . . 83

4.4.3 Numerical Simulation 84
4.5 Moral hazard 91
4.5.1 Input-based payments 91
4.5.2 Input-based payments with uncertainty 94
4.5.3 Output-based payments 96
4.5.4 Output-based payments with uncertainty 100
4.5.5 Discussion 104
4.6 Conclusion 107
4.7 Appendix 109
4.7.1 Derivation of Equation (4.21) 109
4.7.2 Estimation of ES value 110
4.7.3 Crop density of soybeans and corn in Ontario 111
4.7.4 Lemma $4.5.1$ 111
4.7.5 Derivation of Equation (4.46) 111
4.7.6 Derivation of Equation (4.53) 113
4.7.7 Derivation of Equation (4.55) 115
4.7.8 Responsiveness of $\alpha^*(x)$ to changes in x 118
4.7.9 Relationship between α^* and land input when $\psi(x)$ is linear in x 118

5 Conclusion **120**

5.0.1 Limitations 123

References **125**

List of Figures

1.1 Water cycle-forest interactions (Brauman et al. 2007) 9

1.2 The Multi-Barrier Approach (Canadian Council of Ministers of the Environment 2002) 11

2.1 Distribution of service estimates by country 29

2.2 PRISMA 2020 Flow Diagram of Data Collection Process 39

4.1 First-Best and Second-Best Robust Input Allocation 88

4.2 First-Best and Second-Best Robust Payments 89

4.3 Expected Social Welfare Once Distribution Is Realised 90

List of Tables

2.1 Summary of research on the impacts of deforestation on forested watershed ecosystem services . . . 19
2.2 Summary of meta-analyses findings on ecosystem services valuation . . . 21
2.3 Description of explanatory factors . . . 26
2.4 Univariate analysis of economic values (2020 USD/ha/year) across ecosystem service categories, valuation methods, and forest types. . . . 31
2.5 Estimation results for models with forest area, site area, and main and detailed ES classification . . . 33
2.6 Original valuation studies and number of observations. . . . 38

3.1 A summary of forested watershed valuation studies using cross-sectional analysis . 44
3.2 Summary of Dependent and Independent Variables . . . 51
3.3 Spatial Regression Results with Land Use Characteristics . . . 57
3.4 Structural Equation Model Regression Results . . . 62
3.5 Inference on the Mediated Effect for Land Use . . . 63
3.6 Direct, Indirect, and Total Spatial Effects of Land Use on Turbidity at 5km, 10km, and Sub-Sub Drainage Area . . . 67
3.7 Direct, Indirect, and Total Spatial of Determinants of Drinking Water Treatment Cost at 5km, 10km, and Sub-Sub Drainage Area . . . 68

4.1 Summary of Simulation Parameters . . . 86
4.2 Numerical estimates of optimal policy . . . 87
4.3 Optimal Payment Structures under Different Scenarios with Moral Hazard . . . 106

4.4 Structural Equation Model Regression Results 110
4.5 Soybean and Corn Densities by Land Type 111

Chapter 1

Introduction

The 2005 Millennium Ecosystem Assessment (MA), a landmark global initiative to assess the consequences of ecosystem change on society, defines ecosystem services (ES) as "the benefits people obtain from ecosystems" (Millennium Ecosystem Assessment 2005, p. v). The MA emphasises that the sustainable management of a wide range of ecosystems, including natural forests, is often the most cost-effective approach to ensuring the provision of critical services essential for economic growth and human well-being. Today, however, forests face unprecedented pressures from human activity, climate change, and land use change. The Food and Agriculture Organisation (FAO) highlights in the 2024 State of the World's Forests report that forest loss and degradation is continuing at alarming rates, with over 10 million hectares lost annually to deforestation driven primarily by agricultural expansion. Forests play a critical role in regulating watersheds, and so are essential for maintaining water quality, controlling sedimentation, and regulating surface flow and groundwater recharge (Brauman et al. 2007), yet these services are increasingly threatened by unsustainable practices. Should environmental degradation occur, replacing the services forests provide would require significant economic investment. The ES framework has been instrumental in providing a conceptual model by which to understand the costs and benefits of conservation.

While early economists like David Ricardo and Adam Smith recognised that natural resources are essential for production, they focused primarily on land and labour inputs, and explicit attempts

to model the role of ecosystems were historically limited. This oversight was partly because natural resources were ”superabundant” relative to smaller-scale human activity in their time (Costanza and Daly 1992). As human activity has intensified and natural resources have become more scarce, the need has grown for a more structured approach to integrating ES into economic models. The popularisation of the ES framework, particularly following the 2005 MA, marked a turning point for the field of environmental economics. The ”eye-opening metaphor” provided the conceptual tools necessary to incorporate the benefits ecosystems provide into the neoclassical model of economics (Norgaard 2010). To an economist, ecosystems can be characterised as ”renewable natural capital” - ”a stock that yields a flow of valuable goods and services into the future” (Costanza and Daly 1992, p. 38).

Once we recognize that ecosystems provide vital services, it seems intuitive that they should be protected. However, this protection rarely occurs without government intervention, as many ES are public goods - benefits people enjoy without direct market interactions (Costanza et al. 1997). This leads to a critical challenge in decision-making: ecosystem services, not being monetised, are frequently overlooked by policymakers. Pagiola et al. (2004) illustrate the dilemma faced by policymakers: despite knowing the economic importance of ecosystems for a myriad of vital industries, the value of these ES remains largely invisible in national statistics. This invisibility stems from the difficulty in quantifying and monetising ES. Environmental economists have focused on developing methods to value these services since Costanza et al. (1997) famously and controversially estimated the global value of ES at $33 trillion, nearly double the global Gross National Product (GNP) at the time.

One approach to securing the continued provision of ES that is highlighted in the MA is Payments for Ecosystem Services (PES). PES schemes are voluntary transactions where landowners are compensated for conservation in exchange for the provision of ES. Such schemes are defined by four key characteristics: they are voluntary, involve at least one buyer and one seller, and most importantly, are **conditional**. This last feature is critical, requiring that payments are made ”if and only if the service provider continuously secures the provision of that service over time” (Wunder

2005, p. viii). PES schemes typically target four categories of ES: carbon sequestration, biodiversity, landscape beauty, and watershed protection (Wunder 2005). Watershed services, in particular, offer significant benefits to human welfare, though they are often the most difficult to quantify and monitor, adding uncertainty to PES implementation.

Further, Payments for Watershed Services (PWS) schemes are often established without a formal valuation study being conducted beforehand (Porras et al. 2013). In many cases, PWS emerges in environments where there is a clear demand for ES - but driven by government initiatives or top-down arrangements rather than reflecting direct WTP from service beneficiaries (Ojea and Martin-Ortega 2015). Often, the precise economic value of the ES provided is unclear, and payments are assigned based on assumptions about the benefits of conservation, or on the precautionary principle, rather than as a result of concrete valuations (Wunder 2013). The success of PES schemes hinges on ensuring conditionality, which is a central concern in the context of watershed protection. Although PES schemes have gained popularity, there remain significant challenges in proving their efficacy, particularly due to uncertainty surrounding the linkages between forest cover, land use, and water services (Porras et al. 2013; Farley and Costanza 2010). The failure to demonstrate that landowners consistently provide the promised services may create skepticism among service users and threaten the long-term viability of PES programs (Wunder 2005). Conversely, without valuation, there is a risk that payments may be too low to incentivise long-term conservation, or that service users may undervalue the benefits they are receiving.

Thus, valuation can be an important complementary step in PES design, as a way to ensure that payments align with the economic value of the services provided. Challenges in valuation underscore the need for a deeper understanding of the uncertainties that cloud the relationship between watershed ecosystems and the services they provide. This thesis addresses the central problem of uncertainty and conditionality in PWS. Uncertainty with respect to ES arises from many factors including natural environmental variability, data limitations, and unforeseen ecological and behavioural feedbacks (Millennium Ecosystem Assessment 2005). The following chapters explore three key questions: (1) What do we know about forested watershed ES and the determinants

of their value? (2) How can we use valuation to extend our knowledge and reduce uncertainty surrounding the value of these services? (3) Given the inherent uncertainty, how can we design PES contracts to ensure better outcomes?

The second chapter of this thesis advances the field by answering the first question: *What do we know about forested watershed ES and the determinants of their value*? Several researchers have synthesised valuation literature on forested watershed ES, but these efforts have often been limited in scope, focusing primarily on tropical watersheds (e.g., Chomitz and Kumari 1998; Lele 2009). Further, while these reviews provide critical insights into valuation, they have primarily been qualitative, lacking a systematic analysis that can help pinpoint key determinants of between-study variation in monetary estimates of watershed services. More recently, Ojea and Martin-Ortega (2015) carry out a quantitative analysis on forested watershed ES. However, they restrict their dataset to watershed ES in Central and South America, and do not differentiate between ES due to the risk of double-counting and service overlapping, as many of the studies they identify aggregate services or are not explicit about the service measured. However, this is a significant oversight, failing to differentiate between ES can obscure the factors that drive variation in estimates. This chapter fills this gap by synthesising the global valuation literature on forested watershed ES.

Studies in this analysis estimate a wide range of forested watershed ES, with key variables including study characteristics like valuation methodology, site characteristics like forest type or national income, and service characteristics. The Common International Classification of Ecosystem Services (CICES) framework is utilised to categorise the different services provided by forested watersheds. Two sets of hypotheses are tested: (1) The inclusion of specific ES as explanatory variables will significantly explain between-study variation in value estimates, and (2) The differences between services will reflect scientific knowledge about watershed ES. For instance, the forest hydrology literature suggests that certain regulating services, particularly sediment control, are more consistent across contexts, while provisioning services such as surface flow are highly context-dependent (e.g., Bruijnzeel 2004; Hamilton 2008). A meta-regression using a Generalised Least Squares (GLS) specification is applied to identify the drivers of heterogeneity in valuation

studies, with the aim of understanding determinants of value. The findings highlight that service estimates fluctuate depending on end-users and methodology, reflecting underlying uncertainty around ES production. Despite these challenges, the study identifies several commonalities in the literature, such as the tendency for regulating services to generate higher estimates and for per-hectare values to decline as forest size increases, indicating economies of scale in ES provision. Further, estimates have been decreasing with time, in-line with findings from other meta-analyses that methodological refinements in recent decades have improved model precision, resulting in lower but more accurate estimates. PWS payments are found to be significantly lower than ES estimates in the literature, which has practical implications for the design and implementation of such schemes. Finally, increasing specification using the CICES framework greatly increases the explanatory power of the model, indicating that ES are unique drivers behind different service estimates.

The third chapter seeks to address the second question: *How can we use valuation to extend our knowledge and reduce uncertainty surrounding the value of forested watershed ES*? One of the highest-value services provided by forests comes from their natural filtration function, reducing pollution and sedimentation into waterways (Postel and Thompson Jr. 2005). In Canada, where forests and water are abundant, these watershed services are often taken for granted. However, the importance of these services cannot be overstated, especially considering that in the United States and Canada, at least 50-75% of the population depends on forested watersheds to produce high-quality drinking water (Neary and Ice 2009). Despite their importance, the value of these services remains largely unquantified in the Canadian context, particularly with respect to their impact on water treatment costs. International studies in Ethiopia (Mulatu et al. 2021), France (Abildtrup et al. 2013; Fiquepron et al. 2013), South Africa (Gelo and Turpie 2022), and the U.S. (Warziniack et al. 2017) have demonstrated that forests play a crucial role in maintaining water quality by reducing turbidity, sedimentation, and other pollutants.

No such study has been conducted in Canada. The closest comparable study by Price et al. (2017) used Statistics Canada's 2011 Survey of Drinking Water Plants (SDWP) to estimate the

impact of raw water quality degradation (proxied by turbidity) on treatment costs. They did not incorporate land use data, which is critical for understanding the role of forest cover in mitigating water treatment costs. Additionally, no existing valuation studies have modelled the impact of wildfires on treatment costs. This is important when considering watershed services because wildfires drastically alter landscapes, increasing erosion, runoff, and sedimentation into water bodies (Emmerton et al. 2020). As climate change intensifies, the frequency and severity of wildfires are expected to rise, making it crucial to understand how these disturbances impact the economic and ecological functions of watersheds. Ignoring this factor risks underestimating the true costs of managing water resources and preserving ecosystem services in forested areas. For instance, the State of the World's Forests (2024) report highlights how climate change exacerbates wildfire intensity and frequency. In Canada alone, 2023 saw a record-breaking wildfire season, with fires burning 15 million hectares of land - more than seven times the average (Natural Resources Canada 2023).

To fill this gap in our collective knowledge, this chapter carries out a spatial econometric analysis of the relationship between forest cover, water quality, and drinking water treatment costs in Canada. Using data from the 2015 SDWP and two government datasets - the 2015 Land Cover of Canada (LCC) and the 2017 Canada Landsat Disturbance (CanLaD), this study is the first to explicitly link forest cover and wildfires to turbidity and water treatment costs. By merging the three datasets, the analysis models the impact of different land use types (e.g., forest, agriculture, urban) and wildfires on water quality, and estimates how these factors affect the annual costs of drinking water treatment. Two primary hypotheses guide this analysis: (1) Forest cover, relative to agricultural or urban land use, results in significant cost savings in treatment costs by reducing turbidity levels, and (2) Wildfires, which are expected to increase turbidity, significantly raise water treatment costs. The findings from this chapter have major policy implications, particularly for the design of PWS aimed at incentivising forest conservation. By quantifying the monetary benefits of maintaining natural forest cover, this chapter provides a clear economic rationale for incorporating conservation into water management strategies. At the same time, there is evidence

of the risk posed to drinking water treatment facilities as climate change increases the likelihood and intensity of wildfires, which are projected to become more frequent and severe due to rising global temperatures and prolonged drought conditions (IPCC 2023). This underscores the need for management strategies that not only emphasise forest conservation but also incorporate wildfire risk mitigation into watershed protection policies, as highlighted in the State of the World's Forests (2024) report. Such considerations are essential for ensuring the resilience of drinking water supplies and reducing long-term treatment costs in fire-prone regions.

The fourth chapter explores the theoretical underpinnings of PWS contracting to answer the third question: *Given persistent uncertainty, how can we design PES contracts to ensure better outcomes*? Existing literature has examined the design of agri-environmental contracts under adverse selection and moral hazard (e.g., Ozanne and White 2007; White and Hanley 2016). However, most theoretical exercises construct models that do not fully capture the challenges that characterise ES provision. It is generally assumed that principals and landowners have complete information regarding ES production, and any challenges in contracting arise from informational asymmetries between the two parties. This assumption is too strong, particularly in the case of watershed services. Often, PES schemes are created even as the true production function is uncertain (Zabel and Roe 2009; Wunder 2013). Further, as the performance of PES schemes sometimes lags behinds expectations (Brouwer et al. 2011; Wunder et al. 2018; European Court of Auditors 2020), theorists and policymakers are starting to advocate for output-based contracts to ensure conditionality (Schwarz et al. 2008; Herzon et al. 2018). In the theoretical literature, little attention has been given to the aforementioned complexities in contracting for PES. Building on the works of Derissen and Quaas (2013), who model ES provision as a stochastic process, and Li et al. (2023), who address incomplete information in contracting, this chapter integrates uncertainty into the principal-agent framework.

Uncertainty here is defined as "the lack of confidence that a decision-maker has about possible outcomes and/or the probabilities of these outcomes" (Hamel and Bryant 2017, p.2). In contrast to risk, uncertainty is not as easily quantifiable, which complicates decision-making (Berstimas

et al. 2011). The novelty of this chapter lies in its integration of robust optimisation techniques into the design of PES contracts. Drawing on robust optimisation literature (e.g., Carroll 2015, Yu and Kong 2020), a principal-agent model is developed where neither the principal nor landowners know with certainty the benefits generated from conservation, and principals must design a scheme that performs well under their worst-case assumptions about the potential for ecosystems to provide services. The model additionally incorporates adverse selection, where landowners have private information about their costs, and moral hazard, where they are able to shirk from their obligations. Two sets of hypotheses are tested: (1) Uncertainty changes the structure of optimal contracts in PES, and (2) In the presence of moral hazard, contracts with output-based payments allow principals to reduce some of the challenges presented by uncertainty. The results suggest that with input-based payments, principals should be conservative when investing in ES provision. In this case, the challenges of uncertainty are placed on principals, and thus a 'conservativeness' approach (Angelsen 2008) can ensure better PES outcomes given worst-case assumptions about ES hold true. When considering output-based contracts, the burden of uncertainty can be shared between principals and landowners by offering a hybrid contract. Here, the optimal contract includes two components. A fixed payment that compensates landowners for their opportunity costs ensures participation, while a variable component where they receive a fixed share of the value of output ensures compliance. These findings align with the growing literature on the potential of output-based contracts in PES, however, such programs may be difficult to implement in cases where ES output is difficult to quantify or costly to monitor.

1.1 Forested Watersheds and the Provision of Drinking Water

Forests play a critical role in the provision of water-related ES, especially the maintenance of high-quality water for downstream users. While watersheds provide many services, from biodiversity conservation and carbon sequestration to spiritual and recreational uses, arguably one of the most vital is the provisioning of clean water for drinking. This service is directly tied to Sustainable

Development Goal 6 (Clean Water and Sanitation), which is itself foundational to achieving other objectives such as Zero Hunger (SDG 2), Good Health and Well-being (SDG 3), Sustainable Cities and Communities (SDG 11), and Responsible Consumption and Production (SDG 12) (UN Water 2016). Historically, improved water quality has played a critical role in reducing urban mortality. The use of chemically assisted filtration and disinfection accounts for nearly half the decline in total mortality rates in US cities between 1900 and 1936 (Cutler and Miller 2005).

Any ecosystem will influence the water that passes through it, and land use profoundly impacts water quality through its impact on the hydrological cycle, sediment transport, and nutrient loading (Brauman et al. 2007). Forests function as regulators of water flows and filters of contaminants. The interactions between vegetation, soil, and water occur at multiple stages, from precipitation to river discharge, as depicted in Figure 1.1.

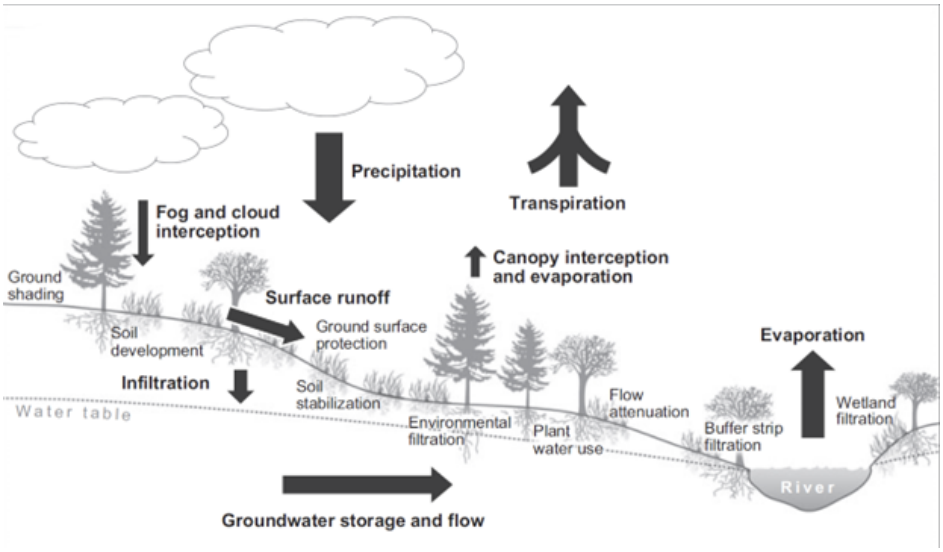


Figure 1.1: Water cycle-forest interactions (Brauman et al. 2007)

The diagram broadly describes the linkages between forests and the hydrological cycle. The protection of drinking water by forested watersheds begins with precipitation. Forest canopies intercept rainfall, reducing the energy of raindrops impacting the soil surface. This process limits erosion by protecting soils from raindrop splash and promotes groundwater infiltration (Ilstedt et al. 2007). Root systems further stabilise the soil and remove nutrients from water, reducing the risk of erosion as well as managing sediment delivery into nearby waterways (Gyssels et al. 2005;

Mayer et al. 2007). Due to vegetation and soil compaction, forests also slow down water in a process known as flow attenuation. By reducing surface runoff, forests further prevent the mobilisation of pollutants (Neary et al. 2009). Riparian buffers are especially important. Vegetation adjacent to streams and rivers act as the final line of defense against contamination, trapping sediment, absorbing excess nutrients, and stabilising soil (Brauman et al. 2007; Mayer et al. 2007). The role of land use in supporting the provisioning of drinking water is well-documented empirically. Studies have shown that upstream forest cover is associated with improved raw water quality and reduced turbidity at drinking water intakes (Ernst et al. 2004). Conversely, land use change typically degrades water quality through increased surface runoff, sedimentation, pesticide loads, and nutrient enrichment (Allan 2004; Foley et al. 2005). These impacts increase the operational and chemical costs of drinking water treatment (Abell et al. 2017).

However, the costs avoided by source water protection can be hard to quantify. There is a misconception that high-quality water requires little to no treatment. All drinking water, regardless of source, requires treatment to mitigate risk. Even the highest-quality groundwater sources must be disinfected to meet minimum public standards such as 4-log (99.99%) virus removal (Emelko et al. 2019). This contributes to hidden inefficiencies and can make establishing causality difficult. Unlike in other economic contexts, where the duplication of services is seen as a sign of inefficient resource use, redundancy is considered optimal when public health is at risk (O'Connor 2002). Nevertheless, the maintenance of water quality by forested watersheds is evidently critical, particularly in North America. In 2004, the American Water Works Association (AWWA) began surveying water professionals in North America regarding key industry concerns. Over 17,000 professionals have since participated. Source water protection consistently ranks among their top priorities, topping the list in 2024 (Flancher 2024). While treatment will always be necessary, it is not sufficient for providing clean drinking water. In Canada, the multi-barrier approach (Figure 1.2) which is the driving principle behind drinking water safety management, calls for robust source water protection in addition to traditional treatment. It was officially recommended in the wake of the 2000 Walkerton tragedy, where contamination from agricultural runoff led to seven deaths

and thousands of illnesses due to E. coli in the municipal water supply (O'Connor 2002). Canadian drinking water policies increasingly prioritise source water protection. Federal and provincial guidelines, e.g., the Guidelines for Canadian Drinking Water Quality (GCDWQ), stress that relying solely on engineered treatment is insufficient, particularly in the face of climate-related threats such as floods, droughts, and wildfires (Emelko and Sham 2014).

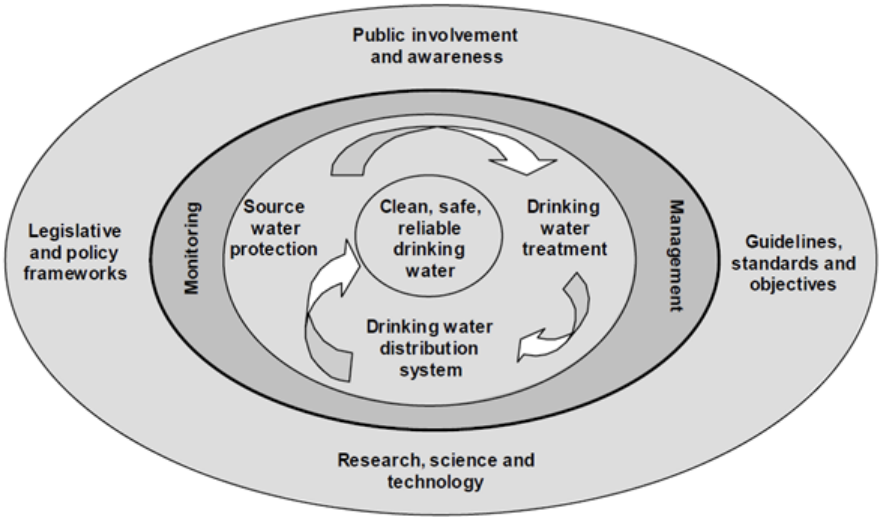


Figure 1.2: The Multi-Barrier Approach (Canadian Council of Ministers of the Environment 2002)

Wildfires represent one of the most severe threats to the water quality maintenance service provided by forested watersheds. Fires alter the flow of these services in many respects. They alter vegetation structure, increase the water repellence of soils (Neary et al. 2009), reduce canopy interception, and expose bare soils to erosion (Emelko et al. 2011). Post-fire catchments face elevated sediment yields and turbidity, which may challenge treatment facilities in the short- and long-term (Hohner et al. 2019). Water treatment facilities not designed for large fluctuations in nutrient loads, heavy metals, or the presence of algal blooms may struggle to adapt (Emelko et al. 2011). Worsening water quality often necessitates additional treatment, and in extreme cases can force a facility to shut down (Hohner et al. 2019). Importantly, the impact on water quality can persist long after the fire has subsided (Bladon et al. 2014). In this context, the multi-barrier approach can also be understood as a climate resilience strategy, especially in regions where wildfires are projected to become more frequent and severe, as is the case for much of Canada

(Wotton et al. 2017).

The complexities of water treatment are often misunderstood. The costs of treating drinking water can be buried in capital investments, while the benefits of redundancies (e.g., multi-intake facilities) are only visible in the case of emergencies. Treatment costs do not scale linearly with input factors. They exhibit step-function characteristics – in large metropolitan systems like Toronto, facilities are designed for regulatory compliance in worst-case scenarios that may never materialise, mandating a one-size-fits-all approach that over-designs or over-treats water in many cases (Ballantyne et al. 2014; Emelko et al. 2019). At first glance, it may seem that drinking water treatment is uniform and strictly regulated across Canada, given that the GCDWQ set national benchmarks for the safety of drinking water and are supported by provincial legislation in many jurisdictions.

While public safety is a non-negotiable requirement, and all utilities must comply with microbial and chemical standards set by the GCDWQ (e.g., Dunn et al. 2014a), there is some heterogeneity in application. For example, out of the 94 guidelines parameters in the GCDWQ, only 16 are applied consistently across all jurisdictions in the country (Dunn et al. 2014b). This regulatory flexibility means that decisions about treatment and infrastructure can also be shaped by broader societal and institutional dimensions. Characteristics of treated water like taste and odour can vary depending on public attitudes. For example, while disinfection is essential for eliminating microbial pathogens, it can also affect taste and produce disinfection byproducts that can pose health risks with long-term exposure (Health Canada 2006). Various stated preference studies have shown that respondents are often willing to pay to reduce exposure to water that tastes of chlorine (e.g., Danish Environmental Protection Agency 2002; Jeuland et al. 2016). In British Columbia, disinfection byproducts are considered a public health concern, and it has been difficult to convince communities of the need for disinfection in the face of public resistance (British Columbia Provincial Health Officer 2001).

Affordability is another critical concern. Facilities in communities with limited fiscal capacity struggle to meet minimum regulatory requirements, as is increasingly the case in Atlantic Canada. Due to an aging and diminishing population, small communities (< 1000 people) in Newfoundland

and Labrador find fully meeting regulatory standards impossible; these communities are also more likely to be subject to long-term boil water advisories (Minnes and Vodden 2017; Ochoo et al. 2017). Utility operators in the rural communities of Nova Scotia and New Brunswick frequently encounter resistance to rate increases and skepticism towards new treatment technologies, even when required to meet regulatory standards (Kot et al. 2011).

Understanding the true effect of forest cover on drinking water quality would ideally require a dataset and model that integrate the biophysical, institutional, and economic dimensions discussed here. The ecological and economic linkages between forested watersheds and the provisioning of clean drinking water are well-established in principle. However, as these discussions highlight, the actual contribution of forests to drinking water quality is obscured by uncertainty, both in terms of how ecosystems function and how communities respond to land disturbances. At the biophysical level, an ideal dataset would include spatial data on upstream land use and track multiple water quality indicators on turbidity, nutrient loads, and microbial pathogens. From an institutional perspective, the dataset and model would need to incorporate how regulation and enforcement vary across jurisdictions in the event of wildfires, algal blooms, and other disturbances. It would also capture economic and behavioural factors that influence operating costs. While many studies have demonstrated associations between forest cover and raw water quality, these relationships remain difficult to generalise due to the absence of this specific information, and modelling approaches will be limited. Uncertainty, where outcomes and their probabilities are unknown, becomes a critical factor.

Forested watersheds contribute to drinking water quality and are a crucial part of protecting communities from health borne risks. The natural filtration and flow-regulating services provided by forests reduce pollutant loads in source water, lowering the burden on built infrastructure. However, the extent to which these benefits are realised depends not only on biophysical processes, but also on the broader socio-political and regulatory context in water resource management. As this thesis will explore, understanding these linkages is essential for evaluating the full value of watershed conservation and designing policy mechanisms like PES that recognise forests not only as

ecological assets, but as providers of essential services to communities.

Chapter 2

Testing the Effect of Ecosystem Service and Land Classification on Global Values of Forested Watershed Ecosystem Services

2.1 Introduction

Forested watersheds are widely considered to play a crucial role in securing a variety of benefits for downstream communities, particularly in terms of water services (e.g., Ovando and Brouwer 2019). The Millennium Ecosystem Assessment (MA) promotes an ecosystem services (ES) framework to recognising the role ecosystems play in human welfare (Ojea et al. 2012; Brandon 2014; Hamilton 2008; Molnar and Kubiswzeski 2012; Mullan 2014). Ecosystem services are the processes that link natural ecosystems to social welfare and well-being (Millennium Ecosystem Assessment 2005), where a given ecological process can impact human welfare through various pathways, e.g., through provisioning services which satisfy the material needs of society like drinking water or fish, or cultural or non-material services like education or recreation. The ES approach has been championed in environmental economics due to its utility for conservation, particularly by allowing policymakers to compare the Total Economic Value (TEV) of different land manage-

ment scenarios in cost-benefit analysis. However, in order to obtain directly comparable benefits of ecosystem conservation to economic growth and development, economic valuation studies are key to assigning monetary estimates to the functions performed by natural ecosystems generating non-market benefits (Pagiola et al. 2004).

The publication of ES valuation studies has increased rapidly in the last decade (Hackbart et al. 2017), indicating a growing recognition of their importance in environmental policymaking and natural resource management. ES valuation is thus a fundamental tool for informed sustainable land use policy, as many of the services provided by forested watersheds do not enter markets before consumption. If only marketed goods are considered in conservation policy, land use change to urban or agricultural may appear to be the economically sound choice more often than economically warranted according to the TEV framework. Consequently, valuation of non-marketed benefits bolsters conservation efforts where they might otherwise be overlooked. According to the 2020 Global Forest Resources Assessment, since 1990, 178 million hectares of forest land have been lost to land use change. Despite these losses, the report also highlights that more forests than ever before are under management plans dedicated to the provision of ES, particularly in the last decade (Food and Agriculture Organisation 2020). Incentive-based programs like Payments for Ecosystem Services (PES) have gained traction as a way to encourage the conservation and sustainable management of forest ecosystems (Wunder et al. 2018). By translating the non-market benefits of forests into financial terms, PES programs make the conservation of forests advantageous compared to other land use alternatives. PES schemes vary considerably in duration and structure, but are commonly short-term contracts (e.g., 5-20 years), with many longer contracts front-loading payments in the early years (Wunder and Alban 2008; Mahanty et al. 2013).

In spite of the growing body of peer-reviewed literature on the economic valuation of forests and the implementation of the ES framework in policy and decision-making, there have been few efforts to synthesise the existing literature on the economic value of forest water services. Synthesis can help determine commonalities in valuation (e.g., determinants of value), and identify knowledge gaps for future research (Taye et al. 2021). Forested watershed valuation studies have

been heterogeneous in method and purpose (Ojea and Martin-Ortega 2015), leading to a wide range of estimated values across socio-economic contexts that may be difficult to compare. Meta-analyses have been carried out for forest ES at the global level (Taye et al. 2021; Acharya et al. 2019), for contingent valuation studies (Barrio and Loureiro 2010), for forest recreation values in Europe (Zandersen and Tol 2005), stated preference studies for non-timber benefits in Scandinavian countries (Lindhjem 2007), and for the Amazon rainforest in Brazil (Brouwer et al. 2022). For water-related services provided by forests, this kind of research synthesis is significantly more limited. The TEEB database contains only 33 observations for tropical watersheds, and 18 for temperate and Mediterranean biomes (Van der Ploeg et al. 2010). There is only one existing meta-analysis on forested watershed services, focusing on tropical forests in Central and South America (Ojea and Martin-Ortega 2015).

This study contributes to the literature on forest ES by conducting the first comprehensive meta-analysis on forested watershed valuation studies globally. We explore two critical aspects of the valuation of forested watershed ES. First, we turn to the forest hydrology literature to identify high-value services provided by forests and use the Common International Classification of Ecosystem Services (CICES) framework to systematically categorise estimates from valuation studies. We test the impact of this detailed categorisation of the ES involved on the explanatory and predictive power of our estimated meta-regression model, using a less detailed and more informal ES categorisation typically used in existing meta-analyses (for example based on the Millenium Ecosystem Assessment), if at all, as a control group. Second, we test sensitivity to scale of the ES involved by investigating the effect of the forested area size in the watershed on the estimated economic values. We compare this to the effect of the overall size of the watershed including other land cover and land use. The latter is already a refinement compared to most meta-analyses of forest ES valuation studies that use the national forest area to test for possible diminishing marginal utility for forest ES. Only Zandersen and Tol (2005) use the forested area size at the study site in their meta-regression model of forest recreation values.

2.2 Literature Review

Forests play a critical role in ES provision, particularly for water-related services (Acharya et al. 2019). Studies have used market and non-market valuation methods to assess forest contributions to hydropower (Kaura et al. 2019; Guo et al. 2000), fisheries (Ranjan 2019; Van Beukering et al. 2003), cultural services (Mueller et al. 2019), and risk mitigation for extreme events like floods and drought (Brookhuis and Hein 2016; Kumar et al. 2006). Market-based approaches typically rely on market prices, avoided cost, or production function methods, while non-market valuation employs stated preference methods, such as contingent valuation and choice experiments, to elicit willingness-to-pay (Abdulkarim et al. 2017; Rai et al. 2015). While meta-analyses on forested watershed valuation have been limited, the large body of forest hydrology literature provides us with expectations for valuation. Table 2.1 summarises the research on the ES impacts of deforestation in forested watersheds. The table focuses on the ecological functions that are most often associated with forests: sedimentation, water quality, flow regulation, surface water provision, and groundwater recharge. The second column illustrates the overall trend or direction of the impacts observed, while the third column highlights the key findings and conclusions that support these trends.

Deforestation generally increases sediment delivery into waterways. Forests work as a natural filter. Clearing forests, especially for practices like slash-and-burn agriculture, exposes soil to erosion, leading to greater sediment into rivers and lakes (Lele 2009). Sedimentation in particular reduces reservoir capacity, which impairs irrigation for agriculture and limit hydropower generation (Chomitz and Kumari 1998). While the effect of forest removal on sedimentation is unambiguous, geological characteristics and catchment size are a major determinant of the magnitude of the effect. For example, in larger, flatter catchments, sediment can settle out before reaching water bodies, resulting in a muted effect compared to smaller, steeper catchments (Bruijnzeel 2004). Relatedly, due to the loss of the filtration function of forests, deforestation worsens water quality. Ernst et al. (2004) show that every 10% increase in forest cover in the source area of a water treatment facilities corresponds to a reduction in chemical treatment costs by 20%.

Table 2.1: Summary of research on the impacts of deforestation on forested watershed ecosystem services

ES	Effect	Conclusions	References
Sediment control	↓	*Increased erosion from conversion to soil sealing (e.g., roads), slash-and-burn cultivation, and plantations translates to more sediment in waterways in small basins, reducing reservoir storage volume and impacting irrigation, hydropower, flood control. Varies by geology and catchment size.*	Chomitz and Kumari (1998) Lele (2009) Bruijnzeel (2004)
Water quality	↓	*Forest cover is a natural filter, trapping sediment and reducing nutrient runoff into waterways. One study on 27 water suppliers in the US found that a 10% increase in forest cover in the source area of a treatment facility decreased treatment and chemical costs by* $\sim$ *20% .*	Hamilton (2008) Ernst et al. (2004)
Flood control	↓	*Tropical forests buffer torrential rainfall, reducing flooding frequency and duration. One study using data from 56 countries found that a 10% decrease in forest area increased flood frequency 4-28%*	Bradshaw et al. (2007) Brandon (2014)
	-	*Deforestation generally not linked to increased peak flows, which cause flooding. Conversion to pasture, scrub land can increase peak flows, but not large-scale floods. Linkage to flooding only at a local level (drainage basins* $<$ *50,000 ha). For watersheds >20,000 km*^{2}*, natural processes more responsible.*	Chomitz and Kumari (1998) Lele (2009) Hamilton (2008) van Dijk et al. (2009)
Surface flow	-	*Deforestation has opposing effects on dry-season flows. Removal of trees increases runoff, decreases absorption into ground. Trees also remove water from soil and transpire it into the air, lowering water table. Net effect ambiguous.*	Chomitz and Kumari (1998) Bruijnzeel (2004)
	-	*Conversion to agriculture increases wet-season flows, changes to dry-season flows context-dependent.*	Bruijnzeel (2004) Lele (2009)
	↓	*Conversion to plantation may reduce total and dry season flows, particularly in the case of eucalyptus tree species.*	Lele (2009)
	↑	*Removal of tree cover increases the total amount of water in streams. Higher yields over the year, greatest increase in the dry season.*	Hamilton (2008)
Groundwater	↓	*Safest protection for groundwater is forest cover on sources. Conversion to plantation reduces groundwater recharge.*	Hamilton (2008) Lele (2009)
	↑	*Planting forests on open land where water table close to surface lowers groundwater*	Hamilton (2008)

Note: ↑ indicates that deforestation is associated with an increase in the ES, while ↓ a reduction. − implies an ambiguous effect.

In contrast, the impact of deforestation on flooding is complex and highly scale-dependent, with mixed evidence regarding its significance. At a local scale with smaller flood events, and in tropical regions, forest loss may lead to increased flood frequency and intensity, as trees normally absorb and slow the runoff from rainfall (Brandon 2014). Bradshaw et al. (2007) find that a 10% reduction in forest area increases flood frequency by up to 28%. However, other studies suggest that the influence of forests on flood events diminishes at larger scales. In larger watersheds and for larger flood events, natural hydrological processes play the dominant role in flooding extent, with little influence from forest cover or deforestation (van Dijk et al. 2009). Again, scale matters; catchment size is an important factor in determining the influence of deforestation on ES.

Finally, there is the role of forests in the provisioning of water. Here, the evidence is again mixed, with divergent impacts depending on the season and land use changes considered. It is now consensus that forests do not universally act as 'sponges', storing water in their canopy, litter, and root systems to slowly release in the dry seasons; they can also function as 'pumps' particularly in the tropical regions where transpiration significantly affects hydrological cycles (Lele 2009). Removing trees or converting forested land to horticulture, grasslands or agriculture can increase run-off in the wet seasons by reducing soil absorption capacity, but this is followed in the dry-season by reduced surface flows (Chomitz and Kumari 1998; Bruijnzeel 2004). Generally, forest cover protects groundwater sources, indicating a beneficial effect on groundwater recharge and quality (Lele 2009). Again, it is crucial to consider other factors. Planting forests on open land with shallow water tables can lower groundwater levels, particularly if the tree species has high water demands or high evapotranspiration rates (Hamilton 2008). These findings suggest that, while the impact of deforestation is determined by scale and site-specific factors, provisioning services tend to be highly context-dependent, particularly relative to regulating and maintenance services.

Table 2.2 presents findings of meta-analyses on the valuation of forest ecosystem services that identify approaches to categorising ES, as well as scale-specific factors that are employed in the literature. A recurring issue for meta-analysis is inconsistency in service categorisation due to

Table 2.2: Summary of meta-analyses findings on ecosystem services valuation

Study	Classification	Service coefficients	Size variable	Coefficient	Transformation	Model
Barrio and Loureiro (2010)	NA	Recreational: Baseline Other: +, sig.	National forest area ($\frac{1}{10,000}$)	-0.673*** – -1.123***	Semi-log	Robust OLS
Brouwer et al. (2022)	TEEB	Habitat for species: Baseline Carbon regulation: +, sig. Water cycling: -, sig. Recreational: +, sig.	Site area	-0.762*** – -0.089	Log-log	GLS
Lindhjem (2007)	NA	NA	NR	+, insig. +, insig.	Linear Double-log	Robust OLS
Ojea and Martin-Ortega (2015)	NA	In-stream: Baseline Extractive: +, sig.	National forest area	-0.0355	Log-log	GLS
Taye et al. (2021)	CICES	Timber/ biomass energy: Baseline Other provisioning: -, sig. Atmospheric regulation: +, insig. Other regulating: +, sig. Cultural: -, insig.	National forest area	-0.18***	Log-log	Robust OLS
Zandersen and Tol (2005)	NA	NA	Forest area in site	-1.49e-07*** – -1.08e-6*** -6.21e-06 – 8.84e-05***	Linear Semi-log	Robust RE GLS

OLS refers to Ordinary Least Squares, GLS to Generalised Least Squares, RE to Random Effects.
NA: Not Applicable, NR: Not Reported.

*** $p < 0.001$.

service overlapping and service ambiguity (Fu et al. 2011; Ojea et al. 2012). However, one noticeable trend is the movement towards leveraging structured ES classifications such as The Economics of Ecosystems and Biodiversity (TEEB) or the Common International Classification of Ecosystem Services (CICES), which aim to standardise categories and improve the measurement and comparison of ecosystem services (van der Ploeg et al. 2010; Haines-Young and Potschin 2012). While the meta-analyses in Table 2.2 do not focus as specifically on water-related services as this study, they illustrate key themes in ES valuation. Across studies, the use of structured classification has emphasised the necessity of distinguishing between services, revealing consistent patterns in how they influence variation in ES value estimates, with higher valuations for regulating services and more variable valuations for provisioning and cultural services (Taye et al. 2021; Brouwer et al. 2022).

Another pressing question is how to incorporate spatial scale. Meta-analyses of forest ES valuation typically include size variables as a proxy for ES stock. In addition to the impact catchment size has on ES flows, larger forest area is expected to be associated with a reduced per hectare monetary value of ES due to diminishing marginal returns: as forested area increases, the value-added of each additional hectare can be expected to decrease. Table 2.2 shows that this expectation is consistently observed in the literature, regardless of the model or transformation applied. However, the spatial scale is generally non-specific. Instead of focusing on site-specific or catchment-level measurements, many studies rely on broader proxies. As reporting on site-specific characteristics is generally low, many studies use national forest area as the proxy variable for stock (e.g., Barrio and Loureiro 2010; Ojea and Martin-Ortega 2015; Taye et al. 2021). Only one study uses the area of the study site, which (Brouwer et al. 2022), and further, only one study uses the *forested area* at the study-site (Zanderson and Tol 2005).

2.3 Econometric model, data collection and database

2.3.1 Econometric model

The basic meta-regression strategy is to employ an Ordinary Least Squares (OLS) specification of the form:

$$y_{it} = \alpha + X_{it}\beta + \epsilon_{it}, Var(\epsilon_{it}) = \sigma^2 I \tag{2.1}$$

Where subscript $i = 1, ..., n$ indicates a unique ES value, and subscript $t = 1, ..., T$ indicates the study. y_{it}, representing ES value, is log-transformed, as are all other continuous variables in the database. The error term in Equation (2.1) is assumed to be independent and identically distributed with mean zero. This assumption is unlikely to hold where many values are drawn from the same study (Nelson and Kennedy 2009). In the presence of heteroscedasticity, a more appropriate model is Generalised Least Squares (GLS):

$$y_{it} = \alpha + X_{it}\beta + \epsilon_{it}, Var(\epsilon_{it}) = \sigma^2 \Omega_{it} \tag{2.2}$$

The GLS specification has one key advantage over OLS, by employing the model in Equation (2.2), we can account for correlation between the error terms in the regression analysis. X_{it} is a matrix of regressors represented by:

$$X_{it} = x_{it}^{ser} + x_{it}^{site} + x_{it}^{study} \tag{2.3}$$

Where $x_{it}^{ser}, x_{it}^{site}, x_{it}^{study}$ refer to service, site, and study characteristics, respectively. In the following section, we present our data collection.

2.3.2 Data collection

A systematic search for watershed valuation studies was carried with a global scope. Studies were identified using EconLit, Scopus, and Google Scholar using combinations of the following key phrases: "watershed services"/"forest water services", "value OR valuation", "market"/"production function"/"avoided cost", "hedonic pricing"/"travel cost", "non market"/"contingent market valuation OR CV"/"choice experiment OR CE". Studies were screened by their abstracts for relevance to the study. 64 studies were identified in this way, supplemented by 10 studies from Ojea and Martin-Ortega (2015). Citation searching yielded another 64 studies.

The initial process was followed by a more rigorous screening process. Benefit transfers studies (e.g., Estoque and Murayama 2012; Aziz 2021) were excluded. Studies were furthermore excluded if reported estimates could not be converted into a standardised unit of measurement ($/ha/yr). This included non-monetary ES valuation studies (e.g., Ekaputra and Wardiman 2021; Stickler et al. 2013). Additionally, studies were only included if they explicitly linked changes in forest cover to changes in the flows of ecosystem services. Some studies valued water services in conjunction with non-water services (Thapa et al. 2020; Ojeda et al. 2008). Where it was not possible to extract the economic value of the water ES, the study was not included. Studies were required to include information on their valuation methodology, and were excluded if the methodological approach was unclear (e.g., Croitoru 2007; Ernst 2004). See the Appendix for a PRISMA flow diagram with a more detailed breakdown of the screening process.

Most valuation studies reported information on site area (typically catchment size, but sometimes the area of a nature reserve or administrative region). Forest cover was only reported in 45 studies. Site area has great significance as it can be a proxy for ES supply, particularly in PES schemes (e.g., Brouwer et al. 2011). However, few primary studies report detailed information on site-specific information like forest cover, thus we record both variables to test the robustness of our findings. Other demand-side characteristics, like income and population density, are also rarely reported in valuation studies (Zandersen and Tol 2005). We extract information on national GDP per capita and population density from the World Bank Development Indicators database to

complement the information collected from studies.

Much consideration has been given to the double-counting problem in the categorisation of forested watershed ES (Ojea et al. 2012; Ninan and Inoue 2013). Here, CICES is used to distinguish between ES. The CICES framework incorporates four levels of classification. At the highest level, CICES distinguishes between Provisioning, Regulating and Maintenance, and Cultural services, in-line with the categories defined by the MA (Haines-Young et al. 2012). Increasing specificity facilitates a more detailed categorisation of ES into Classes, Divisions, and Groups. For instance, water purification for human consumption and the control of sediment into waterways for other uses represent distinct Groups in the CICES framework, despite both stemming from the broader forest function of erosion control. By distinguishing these services at a granular level, we can assess the contributions of natural ecosystems to well-being.

All values were converted to 2020 $USD/ha/yr using the Purchasing Power Parity (PPP) adjusted exchange rate and a cross-country database of inflation rates, both from the World Bank World Development Indicators database. Most estimates were reported as annual values. Monthly values were more commonly reported by stated preference studies (e.g., $/household/month). These estimates were multiplied by twelve to obtain annual values. Some studies reported the net present value of a project over a lifetime, these were converted to annuities. Where multiple estimates were reported in a study, the most conservative estimate was included unless the variation could be linked to an explanatory variable in the database and model. This process resulted in a final database with 119 unique observations from 69 primary valuation studies, with an estimated 1.7 value estimates per study.

2.3.3 Database

Table 3.2 presents an overview of the explanatory variables used in analysis, along with their summary statistics. The majority of the estimates ($\sim 60\%$) measure the economic value of regulating and maintenance services, with the erosion control function of forests being of particular interest in the existing literature. The role of forests in maintaining the chemical conditions of water

for drinking purposes ($chemreg$) and in reducing sedimentation for other purposes (erosion) cover 25% and 22% of the database, respectively. While these services describe the same forest function, they distinguish between beneficiaries. Another 13% of the studies estimate the economic value of hydrological cycle regulation ($hydroreg$), particularly to mitigate extreme events like droughts and floods.

Table 2.3: Description of explanatory factors

Variable Type	**Description**	**Obs.**	**Mean**	**St. dev**	**Min-Max**
Dependent variable					
ES value	Value of ecosystem service (in 2020 $USD/siteha/year).	119	305.45	1,803.97	0.003–19,390.23
Service characteristics					
Regmain	Dummy variable indicating estimation of regulation and maintenance (CICES 2).	113	0.60	0.49	0–1
$erosion$	Dummy variable indicating estimation of erosion control service (CICES 2.2.1.1).	114	0.22	0.42	0–1
$hydroreg$	Dummy variable indicating estimation of hydrological cycle regulation service (CICES 2.2.1.3).	114	0.13	0.34	0–1
$chemreg$	Dummy variable indicating estimation of chemical conditions regulation service (CICES 2.2.5.1).	114	0.25	0.43	0–1
Cult	Dummy variable indicating estimation of cultural value of watershed services (CICES 3).	119	8e-03	0.09	0–1
Prov	Dummy variable indicating estimation of provisioning service (CICES 4).	115	0.37	0.49	0–1
$drinksurface$	Dummy variable indicating estimation of drinking water provisioning service from surface water (CICES 4.2.1.1).	108	0.10	0.30	0–1
$nondrinksurface$	Dummy variable indicating estimation of surface water provisioning for non-drinking purposes (CICES 4.2.1.2).	108	0.06	0.25	0–1
$energysurface$	Dummy variable indicating estimation of surface water provisioning for energy (CICES 4.2.1.3).	108	0.05	0.21	0–1
$drinkground$	Dummy variable indicating estimation of groundwater provisioning for drinking (CICES 4.2.2.1).	108	0.10	0.30	0–1

Continued on next page

Table 2.3 continued from previous page

Variable Type	Description	Obs.	Mean	St. dev	Min-Max
$nondrinkground$	Dummy variable indicating estimation of groundwater provisioning for non-drinking purposes (CICES 4.2.2.2).	108	0.05	0.21	0–1
Study characteristics					
$studyyear$	Time in number of years since study.	119	12.94	8.17	1982–2019
$avoidedcosts$	Dummy variable indicating use of avoided cost methods.	118	0.31	0.46	0–1
$othermarket$	Dummy variable indicating use of other market-based valuation techniques.	118	0.20	0.40	0–1
$nonmarket$	Dummy variable indicating use of stated preference valuation techniques (CVM, CE, or hedonic price).	118	0.29	0.45	0–1
PES	Dummy variable indicating PES payment.	118	0.11	0.31	0–1
$replacecosts$	Dummy variable indicating replacement cost (dam storage).	118	0.13	0.33	0–1
Site characteristics					
$sitearea$	Continuous variable recording the site (catchment, reserve, or administrative region) area recorded in each primary study.	118	420,451.40	884,390.50	138.7–4,563,050
$forestarea$	Continuous variable recording the forest area recorded within the site.	78	200,487.7	384,334.70	105.9–1,675,000
$popden$	Continuous variable, population density at time of valuation in each study.	119	115.70	108.12	3.67–485.24
$GDPpc$	Continuous variable, Gross Domestic Product per capita at time of valuation in each study.	119	15,113.19	16,046.57	1,168.14–60,109.66
$tropical$	Dummy variable indicating study forest is Tropical and Subtropical Moist/Dry Broadleaf or Tropical Forest or Subtropical Conifer Forest.	119	0.59	0.49	0–1
$tempconifer$	Dummy variable indicating study forest is Temperate Conifer Forest.	119	0.15	0.36	0–1
$tempbroadleaf$	Dummy variable indicating study forest is Temperate Broadleaf and Mixed Forest.	119	0.18	0.39	0–1

Continued on next page

Table 2.3 continued from previous page

Variable Type	Description	Obs.	Mean	St. dev	Min-Max
$mediterranean$	Dummy variable indicating study forest is Mediterranean Woodland and Shrub.	110	0.08	0.27	0–1

Another $\sim$ 40% of the studies measure the economic value of forest provisioning services. Twenty-one percent consider surface water provision, while 15% estimate the role forests play in promoting groundwater infiltration. Water provisioning studies sometimes model forests as sponges, retaining water in canopy, leaf litter, and the soil. In this sense, forests can be considered green infrastructure, i.e., an alternative to constructing dams or reservoirs, and valuation studies estimate the water storage potential of forest by quantifying the biophysical capacity of a forest ecosystem to store water without necessarily linking these capacities to specific end-user, or direct economic impacts. This approach emphasises the ecological functions and their potential utility, but stops short of connecting them directly to specific stakeholders (e.g., Morri et al. 2014; Mastrorilli et al. 2018; Xue and Tisdell 2001). Conversely, studies actively link forest cover to changes in streamflow or infiltration, and to socioeconomic outcome variables (e.g., Wu et al. 2021; Guo et al. 2001). The latter approach may be more directly useful to policymakers. Only one study estimates the cultural value of forest water services, estimating the public WTP for the preservation of culturally important springs through forest conservation in Arizona (Mueller et al. 2019).

Market-based approaches are far more popular in valuation studies, with the avoided cost method ($avoidedcosts$) being most often used in 31% of all the studies, e.g., estimating savings in the treatment of drinking water resulting from forest conservation. The $replacecosts$ variable corresponds to studies where dam construction costs are used as a proxy for ES value and covers another 13% of the studies in the database. Other market-based approaches that use market prices (e.g., Morri et al. 2014), the production function method (e.g., Wu et al. 2021), or hedonic pricing are less popular. Few studies use revealed preferences to link changes in forest cover to changes in water services, the only study in the database is a hedonic pricing study carried out in the US, which links changes in forest cover to sediment loadings in water bodies and subsequently to variations in residential property values (Yoo et al. 2014). The non-market valuation methods cover

29% of the studies in the database, and consist primarily of stated preferences approaches. The remainder of the studies cover PES schemes. While PES is not a valuation method per se, payments reflect the price paid for service provision. PES payments are determined by opportunity costs, affordability, and political constraints, and should be expected to fall below economic value estimates (Engel et al. 2008).

Figure 2.1 presents the distribution of service estimates by country. Twelve observations are from the United States and another 12 from China, 13 from the European Union, 6 from India, and 5 from Brazil. The majority of watershed valuation studies focus on tropical forests, while temperate and Mediterranean forests are less studied. Over half of all observations are from Tropical and Subtropical Coniferous Forests or Tropical and Subtropical Moist/Dry Broadleaf Forests. Another 18% and 15% of the studies are from Temperate Broadleaf and Mixed Forest and Temperate Coniferous Forest biomes. Eight percent of the dataset covers Mediterranean Forest.

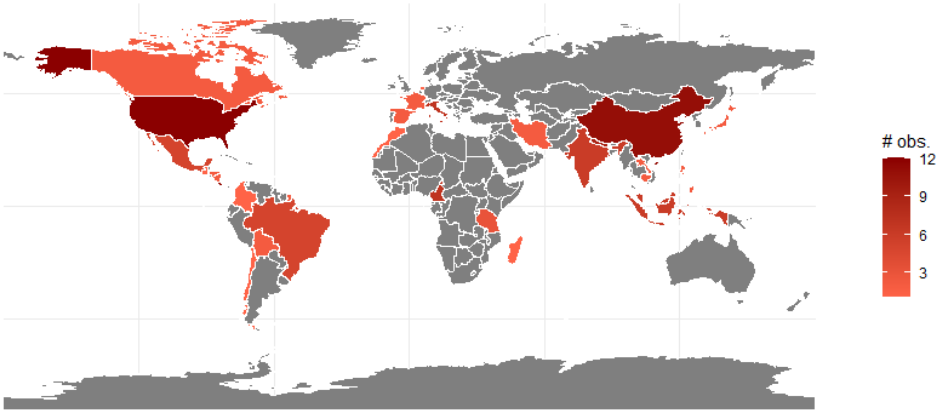


Figure 2.1: Distribution of service estimates by country

2.4 Results

2.4.1 Summary Statistics

Table 3 presents the mean economic values for different ES, grouped according to the good, valuation method, and forest type. We use the non-parametric Kruskal Wallis (KW) test, followed by the Dunn's test for pairwise comparisons, to identify statistical differences in means across groups. The KW test indicates significant differences in ES value across forest types ($\chi^2 = 18.473, p = 0.0003$). The highest mean estimate is derived for $tempbroadleaf$, however the 95% confidence interval indicates there is significant variation in estimates. While tropical forests produce the lowest mean estimate, they also produce relatively speaking the lowest standard error (with a variation coefficient of 40%), and only this variable is associated with a lower 95% confidence bound greater than zero. An important question is to what extent the differences in these values are driven, for example, by differences in scarcity conditions (e.g., deforestation rates) and income levels, since much if not most tropical forest is located in the Global South. Observations from tropical forests are associated with the lowest mean income and the Dunn's test finds that this difference is statistically significant in all pairwise comparisons, i.e., relative to $tempbroadleaf$, $tempconifer$, and $medit$.

The KW test also finds significant differences across the different valuation methods ($\chi^2 = 17.651, p = 0.001$). The mean estimate is highest for the $replacecosts$ category. Dunn's test reinforces this finding, $replacecosts$ estimates are statistically different from every other category. The prices paid in PES schemes are lower at a statistically significant level, while the differences between the other valuation approaches (specify between brackets which ones) are not statistically significant ($\chi^2 = 17.651, p = 0.7412$). This finding is replicated by the Dunn's test. Repeating the exercise for specific ecosystem services does not reveal statistically significant differences across categories, not between the main categories and also not between most detailed ES categories. In the latter case, only $erosion$, $chemreg$, and $drinksurface$ report a confidence interval with a lower-bound greater than zero, highlighting that, despite higher mean values observed in some

provisioning categories, the variations are not driven significantly by ES type.

Table 2.4: Univariate analysis of economic values (2020 USD/ha/year) across ecosystem service categories, valuation methods, and forest types.

	Mean	95% conf. interval	N
Aggregate value for all ES	305.45 (165.37)	-22.03–632.92	119
Forest type			
tropical	80.84 (32.02)	16.96–144.73	70
tempconifer	149.83 (75.06)	-8.54–308.19	18
tempbroadleaf	1007.62 (876.23)	-814.60–2829.85	22
mediterranean	647.17 (306.19)	-58.90–1353.23	9
Main ecosystem service categories			
Regmain	133.48.6 (39.20)	55.24–211.72	68
Cult	3.8	–	1
Prov	556.45 (451.35)	-354.51–1467.31	43
Detailed ecosystem service categories			
erosion	217.27 (94.89)	21.42–413.13	25
hydroreg	77.02 (40.72)	-10.30–164.35	15
chemreg	62.96 (26.82)	7.92–117.98	28
drinksurface	22.00 (7.91)	4.37–39.63	11
nondrinksurface	33.58 (21.34)	-18.38–85.54	7
energysurface	48.10 (17.86)	-1.49–97.68	5
drinkground	175.54 (139.31)	-134.87–485.94	11
nondrinkground	495.49 (355.51)	-491.58–1482.55	5
Valuation method			
avoidedcosts	70.27 (28.37)	12.70–127.85	36
othermarket	146.90 (83.82)	-26.49–320.29	24
nonmarket	110.71 (42.52)	24.18-197.24	34
PES	11.9 (7.39)	-4.20–28.01	13
replacecosts	1782.48 (1271.38)	-944.37–4509.33	11

2.4.2 Estimated meta-regression models

The Breusch-Pagan test on the OLS model residuals consistently confirms the presence of heteroscedasticity in the sample, thus Table 2.4 presents the findings of four different models using the feasible GLS procedure in Stata 13. Models (1) and (2) regress the economic values of forest watershed services on the size of the forested area supplying the ES. In Models (3) and (4), the size of the case study area is used, including forest land and other land cover and uses. Many studies do not report the actual size of the forest supplying the ES. As a result, Models (1) and (2) are based on much less observations than Models (3) and (4). The number of observations drops by over a third if we include the actual forest size as a regressor in the models instead of the size of the case study area. The difference between Models (1) and (2) and Models (3) and (4) is furthermore that the first regression models, i.e., Models (1) and (3), use the broader classification of ES in CICES, while Models (2) and (4) include the more detailed CICES classification of ES.

There is often concern of correlation between service characteristics and valuation methods (Brouwer et al. 2022). A significantly high correlation is only detected between $surfacedrink$ and $market$ ($r = 0.323$). All other pairwise tests indicate correlation coefficients of less than 0.3. All models include site- and study-specific characteristics like forest type, valuation method, and GDP per capita. While population density was recorded as a proxy for service demand, its inclusion resulted in an extremely high VIF, and was therefore excluded from the models presented in Table 2.4. $studyyear$ also has a significant positive coefficient in all 4 models, indicating that studies are generating more conservative (i.e., lower) estimates with time. This may correspond to findings in other research that suggests that methodological refinements in valuation since the early 1990s have resulted in more accurate, but at the same time also more moderate values for ES (Ojea et al. 2010). GDP per capita also has as expected consistently a significant positive effect in each model, indicating that ES values are higher in studies with higher income populations. Interestingly, temperate and Mediterranean forest types generate overall significantly higher values for forested watershed services than tropical forests (the baseline category). However, the effect of specific forest types varies across the different models. For instance, temperate climate conifer

Table 2.5: Estimation results for models with forest area, site area, and main and detailed ES classification

Variable	Model (1)	Model (2)	Model (3)	Model (4)
$constant$	4.176***	2.279**	1.285*	-0.608
	(1.132)	(1.222)	(1.126)	(0.964)
$studyyear$	0.047***	0.091***	0.011	0.031***
	(0.011)	(0.015)	(0.010)	(0.010)
Area size				
$forestarea$	-0.184***	-0.208***		
	(0.044)	(0.039)		
$sitearea$			-0.223***	-0.207***
			(0.042)	(0.035)
Main ecosystem services				
$cult$	-0.733	0.145	-0.885	-0.511
	(0.978)	(0.770)	(1.329)	(0.878)
$prov$	-0.811**		-0.358***	
	(0.175)		(0.173)	
Detailed ecosystem services				
$erosion$		0.938**		1.055***
		(0.320)		(0.156)
$hydroreg$		-0.534		-0.799***
		(0.359)		(0.225)
$drinksurface$		-0.755***		-0.263
		(0.235)		(0.203)
$nondrinksurface$		-2.281***		-0.955***
		(0.441)		(0.326)
$energysurface$		1.448***		1.608***
		(0.522)		(0.271)
$drinkground$		-0.355		-0.605***
		(0.271)		(0.174)
$nondrinkground$		1.021**		0.568
		(0.440)		(0.450)
Valuation method				
$nonmarket$	-1.132***	-0.519**	-0.666***	-0.318*
	(0.196)	(0.263)	(0.219)	(0.187)
$othermarket$	-0.334	-0.533**	0.374***	-0.159
	(0.254)	(0.270)	(0.125)	(0.215)
PES	-2.816***	-2.472***	-1.499***	-2.023***
	(0.262)	(0.262)	(0.292)	(0.209)
$replacecosts$	1.971***	1.937***	2.599***	2.842***
	(0.448)	(0.395)	(0.488)	(0.339)
Type of forest				
$tempconifer$	1.204***	0.216	0.423*	0.145
	(0.389)	(0.342)	(0.245)	(0.226)
$tempbroadleaf$	1.287***	0.946***	0.391	0.760***
	(0.280)	(0.220)	(0.239)	(0.205)
$mediterranean$	0.990*	1.140***	-0.022	0.215
	(0.564)	(0.390)	(0.679)	(0.687)
$GDPpc$	0.040***	0.188**	0.188**	0.385***
	(0.082)	(0.085)	(0.059)	(0.078)
Pseudo-R^2	0.2437	0.3706	0.1627	0.2360
MAPE	1.4907	1.2166	1.5929	1.3710
Root MSE	1.9602	1.6405	2.0459	1.8070
N	75	71	114	107
No. groups	45	42	67	62

$^{*}p < 0.05$, $^{**}p < 0.01$, $^{***}p < 0.001$.

forests produce significantly higher ES values than tropical forests in Models (1) and (3) based on the more generic main ES classification, whereas Mediterranean forests yield significantly higher ES values than tropical forests in Models (1) and (2) only, i.e., as long as the models account for the size of the forest area only.

Using the costs of reservoir construction as a proxy for the hydrological ES values of forested watersheds generates significantly higher estimates than the baseline category ($avoidedcosts$) in all four models, whereas non-market and PES generate significantly lower values. The impact of other market-based methods (production function, market price, hedonic pricing) is somewhat mixed, and only significant in Models (2) and (3) with different signs. This may be due to the fact that the other market-based methods include a wide mix of valuation approaches, ranging from the use of existing market prices for priced water services such as drinking water supply to hedonic pricing of land values in a forested watershed. Overall, these findings indicate that methodological choices for valuation approaches can significantly influence ES estimates. Using reservoir construction costs as a proxy for ES value leverages the concept of natural ecosystems as 'green infrastructure', providing the same services as human-made capital. The fact that different valuation techniques generate significantly different estimates indicates a one-size-fits-all approach may not be appropriate for ES. For example, the findings of a non-market valuation study are, as expected, not comparable to market-based studies.

However, the fact that PES payments generate on average and consistently across all 4 models not only lower value estimates than avoided costs (the baseline category), but also have significantly lower coefficient estimates than non-market valuation approaches to estimate the economic value of ES might suggest three things. Either that such conservation schemes do not fully capture all the benefits provided by ES when their main aim is to protect the overall integrity of the watershed, that existing PES schemes focus on very specific use-related hydrological ES only, such as securing drinking water supply or preventing soil erosion to reduce sedimentation in downstream hydropower reservoirs, or that PES schemes are on average economically speaking very efficient ways of ES preservation with payout costs significantly lower than the total economic value of the

ES involved.

Turning to the coefficients associated with area size, one of the main hypotheses tested in this study, these are as expected always negative and significant. This can be interpreted in the following two ways. First of all, the negative relationship reflects the findings in forest hydrology, that land use changes driving changes in the provision and quality of watershed services typically have a smaller impact on the flow of ES in larger catchments than in smaller catchments. Secondly, the negative coefficient associated with forest area in Models (1) and (2) may also reflect a fundamental trait of economic goods, i.e. the law of diminishing marginal utility. As forest size increases, the benefits from an additional hectare protected fall (Barrio and Loureiro 2010). Interestingly, the effect of forest area size in Model (1) using a broad classification of ES is slightly smaller than the same effect in Model (2) using a more refined and detailed classification of ES. This is even more so the case when comparing Model (1) and Model (3), which both use the broad ES classification, but where the use of the case study area size in Model (3) may overestimate the effect of the stock of forest. This difference in coefficient estimates, and hence overestimation of the size effect, disappears if we apply the more detailed ES classifications in Models (2) and (4). Note also that the two models with the specific forest area size have a higher explanatory power than those with case study area size. Comparing Models (1) to (3) and (2) to (4), we find a 50% increase in explanatory power (pseudo-R^2) with the forest area size variable. Forest cover is a better proxy for ES stock than catchment area, highlighting forest integrity over catchment size in terms of service provision, and the necessity of reporting on site-specific characteristics by valuation studies to facilitate comparisons between sites. Further, we include the relative mean absolute prediction error (MAPE), a measure of predictive power based on the leave-one-out resampling procedure. A lower MAPE for Models (1) and (2) relative to (3) and (4) also indicates a higher predictive power for the meta-regression models including forest cover.

Finally, turning to the ES characteristics, the data validates our expectation that provisioning services yield lower estimates than regulating and maintenance services at a statistically significant level. Although alternative land use is not included in the dataset, the forest hydrology litera-

ture (e.g. Bruijnzeel 2004; Hamilton 2008) suggests that provisioning services are more sensitive to local contexts, while regulating services like water purification and sediment control provide broader, more universally recognized benefits. The more detailed CICES categorisation in Models (2) and (4) aids in determining high-value services. Relative to the baseline ($chemreg$), $erosion$ and $energysurface$ produce consistently higher estimates, while $nondrinksurface$ estimates are significantly lower. This finding indicates that end-use matters. $chemreg$ and $erosion$ are differentiated not by forest function, but by beneficiary, i.e. the general public versus other users such as hydropower facilities. The same is true for provisioning categories, where surface flows for energy generate high values, but surface flows for drinking and non-drinking purposes are not as valuable. Again, the inclusion of more specific service categories increases the explanatory and predictive power of the estimated meta-regression models, where the highest pseudo-R^2 is for Model (2).

2.5 Conclusions

This is the first study to synthesise the global literature on forested watershed valuations. While the literature on the valuation of watershed ecosystem services has grown exponentially over the past decades, this study shows a dire need for improved international coherence and consistency of their definition and measurement, in particular the flow of the specific services involved from existing stands or stocks of forest, as measured through their area size. Although forest conservation is an increasing priority and ES valuation has become a critical tool in informing policy, there remains uncertainty driven by a diffuse and heterogeneous literature. We employ a meta-regression model to formalise the relationship between forested watershed ES estimates and various specific site, study, and service characteristics to support land use policy.

We establish significant links between ES values and key service characteristics by utilising the CICES framework. This significantly influences the explanatory and predictive power of the estimated meta-regression models, and affects at the same time the effect of the stock of supply on the ES values involved, as measured through either forested land or case study area size. We

find that the values of the specific services involved are, as expected, highly dependent on their users, whilst controlling for the impact of different valuation methods, where the highest estimates are found for energy-related (hydropower) services. Studies using the avoided cost of reservoir construction as a proxy for forested watershed services generate significantly higher values than other methods, while non-market valuation methods produce the lowest ES values. PES schemes yield consistently lower ES values. We find, in-line with the existing synthesising literature on ES valuation such as meta-analyses, that there is an urgent need for improved reporting of ES and site-specific characteristics like income, forest area, and the population of beneficiaries. The lack of detailed data and information in the original studies has important consequences for our ability to explain between-study variation and produce reliable predictions of ES values for future use in benefit-cost analysis to support integrated watershed policy and decision-making.

2.6 Appendix

2.6.1 List of studies

Table 2.6: Original valuation studies and number of observations.

Study number	Study	Country	Obs.
1	Abdulkarim et al. (2017)	Malaysia	2
2	Abildtrup et al. (2013)	France	1
3	Acharya et al. (2021)	Nepal	4
4	Amponin et al. (2007)	Philippines	1
5	Arias et al. (2011)	Cambodia	1
6	Ashagre et al. (2018)	Tanzania	1
7	Barkmann et al. (2008)	Indonesia	1
8	Bernard et al. (2009)	Costa Rica	3
9	Bhandari et al. (2016)	Nepal	1
10	Biao et al. (2010)	China	3
11	Brookhuis and Hein (2016)	Trinidad and Tobago	1
12	Brooks et al. (1982)	Morocco	1
13	Castillo-Eguskitza et al. (2019)	Spain	1
14	Chaikaew et al. (2017)	United States	1
15	Das et al. (2019)	India/Nepal	2
16	de Brito et al. (2018)	Brazil	2
17	Elias et al. (2014)	United States	1
18	Fu et al. (2014)	China	1
19	Guo et al. (2000)	China	1
20	Guo et al. (2001)	China	3
21	Guo et al. (2007)	China	2
22	Häyhä et al. (2015)	Italy	1
23	Hein (2011)	Netherlands	1
24	Hérivaux and Grémont (2019)	France	2
25	Hodgson and Dixon (1988)	Philippines	1
26	Holmes et al. (2004)	United States	1
27	Honey-Rosés et al. (2017)	Spain	1
28	Huang et al. (2009)	Indonesia	1
29	Jaina et al. (2017)	India	1
30	Johnson and Baltodano (2004)	Nicaragua	1
31	Kaura et al. (2019)	Cambodia	2
32	Kibria et al. (2017)	Cambodia	2
33	Kosoy et al. (2007)	Honduras/Costa Rica/Nicaragua	4
34	Kumari (1995)	Malaysia	1
35	Kramer et al. (1997)	Madagascar	1
36	Liu et al. (2019)	China	1
37	Lopa et al. (2012)	Tanzania	2
38	Mastrorilli et al. (2018)	Italy	2
39	McGinnis et al. (2021)	Mexico	4
40	Mokondoko et al. (2016)	Mexico	1
41	Moreno-Sanchez et al. (2012)	Colombia	1
42	Morri et al. (2014)	Italy	4
43	Mueller et al. (2019)	United States	3
44	Ninan and Kontoleon (2016)	India	1
45	Ninan and Inoue (2013)	Japan	2
46	Núñez et al. (2006)	Chile	1

Continued on next page

Table 2.6 continued from previous page

Study number	Study	Country	Obs.
47	Pagiola (2008)	Costa Rica	5
48	Pan et al. (2022)	Canada	1
49	Pattanayak and Kramer (2001)	Indonesia	1
50	Piaggio and Siikamäki (2021)	Costa Rica	1
51	Rai et al. (2015)	Nepal	2
52	Ranjan (2019)	India	1
53	Richards et al. (2015)	Brazil	1
54	Rosales et al. (2005)	Laos	2
55	Ruitenbeek (1992)	Cameroon	2
56	Shahwahid et al. (1997)	Malaysia	1
57	Shultz and Soli (2007)	Bolivia	2
58	Shojaie Shami et al. (2021)	Iran	2
59	Simonit et al. (2015)	United States	1
60	Singh and Mishra (2014)	India	2
61	Trenholm et al. (2013)	Canada	1
62	Van Beukering et al. (2003)	Indonesia	3
63	Van Hecken et al. (2012)	Nicaragua	1
64	Vincent et al. (2016)	Malaysia	1
65	Wu et al. (2021)	Brazil	2
66	Xue and Tisdell (2001)	China	1
67	Yaron (2001)	Cameroon	5
68	Yoo et al. (2014)	United States	5

2.6.2 PRISMA 2020 Flow Diagram of Data Collection Process

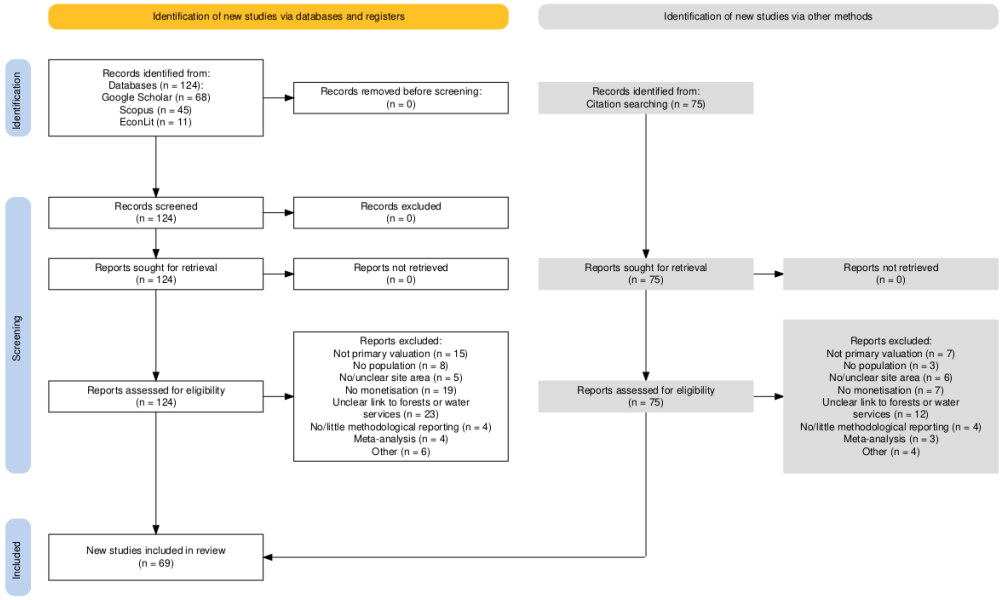


Figure 2.2: PRISMA 2020 Flow Diagram of Data Collection Process

Chapter 3

Estimating The Value of land use Choices on Drinking Water Treatment Costs: A Spatial Regression Approach

3.1 Introduction

The increasing demand for drinking water amidst escalating pressure from human settlement and climate change underscores a critical need for cost-effective water management strategies. Forests are a significant force in supporting water management through their role in the hydrological cycle. Perhaps their most critical service is in maintaining the quality of source water. Forest floors are able to support high levels of organic matter and have extensive root networks which allow them to act as natural filters, reducing surface erosion and trapping pollutants from runoff and baseflow before they reach water bodies like lakes and rivers (Postel and Thompson Jr 2005). The case of the Catskills-Delaware watershed, which received considerable public attention, demonstrates that the costs avoided by not having to construct a filtration system can be extremely high, with New York City saving $6 billion in initial construction costs and $300 million annually in operating expenses by preserving forest cover in the surrounding catchment (Postel and Mastny 2005). The

best quality of fresh water in North America is thought to originate in forested watersheds: 50-75% of Americans receive their water from forested land, and this number is likely higher in Canada (Neary and Ice 2009).

The problem of drinking water provision may not seem as relevant to Canada at first glance, home to an ample 9% of the world's renewable freshwater resources with vast swathes of forestland (Davies and Mazdumer 2004). However, most drinking water sources in the country are close to the communities they supply, and are exposed to the risks associated with human settlement, particularly increased nutrient and chemical runoff in waterways (Dudley and Stolton 2003; Davies and Mazumder 2004). Further, forests in Canada are especially vulnerable to the risks associated with climate change. Increasing temperatures and associated changes in fuel moisture are anticipated to increase the number of days where wildfires are likely, as are the number of days where fire intensities exceed current suppression resources (Wotton et al. 2017). In 2020, approximately 200,000 hectares of forestland was lost to burning in Canada, in 2021 this figure was at approximately 4,300,000 (Canadian Council of Forest Ministers - Conseil canadien des ministres des foreˆts 2020). Wildfires can drastically impact source water quality in watersheds (Emelko et al. 2011; Emmerton et al. 2020). Human activity continues to degrade the natural environment, and despite growing interest in "nature-based solutions" to societal challenges like drinking water provision, no previous watershed valuation study has considered the impact of wildfires on drinking water treatment costs. This raises a central question for this study: how do forests and forested watersheds contribute to drinking water treatment cost savings, especially when forest ecosystems are increasingly threatened by pressures like wildfires?

Similar forested watershed service valuation studies have been carried out in international contexts, but this research is limited in Canada. Such analyses generally take one of two approaches to assigning a monetary value to forests. Some studies (e.g., Warziniack et al. 2017; Pan et al. 2022) *explicitly* model raw water as a function of land use and treatment cost or water price as a function of raw water quality in a two-stage approach to ecosystem service valuation. Others (e.g., Abildtrup et al. 2013; Fiquepron et al. 2013; Mulatu et al. 2021) take the *implicit* approach,

regressing costs or price on land use and interpreting the impact of forest as going through the raw water quality pathway. This distinction is relevant because it affects how accurately we capture the different ways that forests contribute to water services. The explicit approach accounts for specific causal pathways (i.e., first estimating the impact of land use on water quality, then on treatment costs), allowing for clearer insights into how land use change affects treatment costs. In contrast, if the implicit approach is misspecified, it may not accurately capture the benefits forests provide. Land use impacts water services through many pathways, and the literature thus far has not explored how this may impact ecosystem services valuation.

In Canada, Price et al. (2017) use treatment process data available in the 2011 Survey of Drinking Water Plants (SDWP) to analyse the relationship between raw water quality and treatment costs using a stochastic frontier analysis. However, their regression specification makes no explicit link between land use and forest cover in Canada and raw water quality or drinking water treatment. Pan et al. (2022) investigate the relationship between water treatment cost and forest cover in Ontario, but lack explicit data on drinking water facilities; they employ municipal water rates and drinking water incidents as proxies for treatment cost and raw water quality, respectively, to study the impact of forest cover on adverse water quality incidents. They introduce a two-stage model, accounting for spatial auto-correlation and endogeneity, and estimate the water treatment cost savings resulting from forest cover to be between $111 and $148 per hectare. One significant contribution of their work, and a common oversight in the field, is the consideration of multiple pathways for land use to impact treatment costs, including infrastructure capacity and source water types. Generally, studies assume land use only influences drinking water treatment costs through an effect on raw water quality. However, this approach overlooks other relationships between land use and drinking water. For example, there is significant variation in water quality standards across Canada, and urban areas of the country may be more able to meet existing regulations than rural regions due to their more developed infrastructure (Hill et al. 2008). Including direct and indirect pathways makes inference more complex, and previous studies have neglected to incorporate these considerations in their analysis.

This study introduces a novel approach by employing a regression model that accounts for spatial auto-correlation, the impact of wildfires, and multiple pathways through which land use can impact drinking water treatment costs, contributing to our knowledge of the role of forest ecosystems in providing valuable services and informing policymakers on the viability of Payments for Ecosystem Services (PES) mechanisms for water resources management in Canada. To our knowledge, this is the first study in the field to consider the latter two effects in analysis. The remaining sections are organised as follows. Section 3.2 is a summary of the literature on water-related ecosystem services provided by forests. Section 3.3 presents the modelling framework employed in this study, while Section 3.4 describes the data. Section 3.5 provides our findings and a discussion, followed by our conclusion in Section 3.6.

3.2 Literature Review

There has been a sustained effort in better understanding the relationship between ecosystems and human communities and the relationship between forest cover and water-related ecosystem services (ES) is no exception. However, few studies explore the economics of natural ecosystems, particularly in Canada. Warziniack et al. (2017) employ a two-stage approach to study the relationship between forest cover and drinking water treatment costs using data from 37 treatment plants across the United States. In the first stage, raw water quality is modelled as a function of natural ecosystem attributes, including drainage area size, forest proportion, proportion of land that is developed (residential, commercial, and industrial use), and animal units per square kilometer in a watershed. In the second stage, treatment costs at a facility are regressed on predicted raw water quality and other treatment plant characteristics (e.g., volume processed, dummies indicating whether conventional, direct, and advanced treatment technologies are used). They find that increasing forest cover in a watershed by one percent reduces turbidity by three, and that a one percent increase in turbidity further increases treatment costs by 0.19%, establishing a clear link between forest cover and water treatment costs via the impact forests have on raw water quality.

Table 3.1: A summary of forested watershed valuation studies using cross-sectional analysis

Study	**Country**	**Cost variable**	**Spatial Scale**	**Model type**	**Include Wildfires**	**Value**	**Explicit/Implicit**
Abildtrup et al. (2013)	France	Water price	Administrative	SM/IV	No	€99-€138/ha/yr	Implicit
Fiquepron et al. (2013)	France	Water price	Administrative	OLS	No	€22/ha/yr	Implicit
Vincent et al. (2016)	Malaysia	Operating costs	Catchment	OLS	No	RM67.75/ha/yr	Implicit
Warziniack et al. (2017)	US	Chemical costs	Catchment	SEM	No	[not estimated]	Explicit
Mulatu et al. (2021)	Ethiopia	Chemical costs	Buffers	OLS	No	[not estimated]	Implicit
Lopes et al. (2019)	Portugal	Operating costs	Buffers	OLS	No	[not estimated]	Implicit
Pan et al. (2022)	Canada	Water price	Catchment	SM/IV	No	$111–$148/ha/yr	Explicit
Gelo and Turpie (2022)	South Africa	Chemical costs	Catchment	SEM	No	R310.60/ha/yr	Implicit

Notes: SEM = Simultaneous Equation Model; SM/IV = Spatial Model with Instrumental Variables; OLS = Ordinary Least Squares.

Thus, a 1% increase in forest cover is associated with a 0.57% decrease in treatment costs through turbidity. They model forest cover as impacting treatment costs through changing the quality of raw water at a treatment facility, a mediation analysis where turbidity and total organic carbon (TOC) are tested as mediators. However, the researchers neglect two major considerations in their analysis. They model forest cover as impacting treatment costs through changing the quality of raw water at a treatment facility, a mediation analysis where turbidity and total organic carbon (TOC) are tested as mediators, but they neglect to carry out inference on their estimated effect. The coefficient associated with the impact of forest cover on turbidity is statistically significant, as is the coefficient estimating the impact of turbidity on treatment costs. That these two coefficients are independently significant does not mean that the joint effect is significant. Further, the researchers neglect to consider spatial relationships between treatment facilities and endogeneity in their model specification.

Abildtrup et al. (2013) estimate with a regression model that increased forest cover reduces water rates in the Vosges region of France, providing households savings of €99-€138 per hectare. They build a spatial lag and error model to account for technical spillovers between neighbouring administrative divisions (e.g., the diffusion of knowledge or technical skills where it is possible for treatment facilities to share specialised labour). They further employ a two-stage modelling approach to account for endogeneity between land cover and raw water quality, and endogeneity between the volume of processed water and prices. Raw water quality is represented as a function of land use choices in an administrative region and its neighbours, the impact of forest cover on water rates is interpreted as a result of its impact on raw water quality.

Vincent et al. (2016) utilise panel data on treatment facilities in the state of Perak, Malaysia to run a regression studying the impact of forest cover directly on treatment costs. They include treatment facility-level fixed effects to control for unobserved relationships between land use and treatment costs, however they do not explicitly model for the impact of forest cover on raw water quality, nor do they account for spatial relationships between neighbouring plants. One major contribution is the inclusion of multiple forest categories to account for ecosystem heterogeneity

- unsurprisingly, virgin (undisturbed) forests are found to play a greater role in water purification than logged forests. Further, the magnitude of the purification service value decreases as the size of the catchment area increases. For the six treatment plants with large catchment areas (>400,000 hectares), the purification service is estimated at RM 1.81 per hectare per year, for the 35 plants that are smaller (<51,000 hectares), the value is estimated at RM 67.75 per hectare per year. Fiquepron et al. (2013) carry out a statistical analysis on the relationship between forest cover and water treatment costs by again modelling raw water quality (represented by indicators for pesticides and nitrates) as a function of land use choice. Forests are again found to reduce water prices for consumers through their impact on raw water quality.

3.3 Methodology

The objective of this study is to estimate the impact of different land use types on drinking water treatment costs through its impact on raw water quality. We address the challenge of the interplay between land use, raw water quality and drinking water treatment costs by employing mediation analysis, which examines how the relationship between two variables, Z (land use) and y (treatment costs), is *mediated* by a third variable, r (raw water quality) (MacKinnon and Webb 2017). The problem can be confounded by other pathways, e.g., if Z impacts y directly outside of r (referred to here as a residual effect). We test sets of models that regress variable treatment costs of treatment facilities on land use and raw water quality characteristics. The 2015 SDWP includes two primary cost categories: operating and maintenance costs, and capital expenditures. The former is of interest to our study, as the latter is predominantly associated with upfront costs for development of infrastructure (e.g., construction of new facilities, upgrading of existing facilities, acquisition of new water sources, etc.), and thus are assumed fixed in the short-run. Our initial model takes the form of the following structural equation model (SEM) specification:

$$y_i = r_i\beta_r + Z_i\beta'_Z + X_i\beta_x + u_{y,i}; \mathbb{E}[u_{y,i}|r_i, X_i, Z_i] = 0 \tag{3.1}$$

$$r_i = Z_i\beta_z + u_{r,i}; \mathbb{E}[u_{r,i}|Z_i] = 0 \tag{3.2}$$

Equations (3.1) and (3.2) describe Model 1. The dependent variable in Equation (3.1) is the natural logarithm of variable treatment costs *per capita* (all continuous variables excluding proportions are log-transformed). β_r and β_x represent the coefficients associated with r (raw water quality) and X (other covariates), respectively.

Here, $\beta_Z^{'}$ represents the *residual effect* of land use on drinking water treatment costs, while the *mediated effect* is characterised by the product $\beta_Z\beta_r$. The sum $\beta_Z\beta_r + \beta_z^{'}$ is the total effect. The focus of this study is the magnitude and significance of the mediated effect of land use on treatment costs through turbidity as a mediator, although we are also interested in testing for a residual effect. Inference of the mediated effect requires estimation of the variance of the product $\hat{\beta}_Z\hat{\beta}_r$. Given the coefficients $\hat{\beta}_Z$ and $\hat{\beta}_r$ with variances $\hat{\sigma}^2_{\hat{\beta}_Z}$ and $\hat{\sigma}^2_{\beta_r}$ respectively, the most commonly used estimator for the variance is:

$$\hat{\sigma}^2_{\hat{\beta}_Z\hat{\beta}_r} = \hat{\beta}_Z^2\hat{\sigma}^2_{\hat{\beta}_r} + \hat{\beta}_r^2\hat{\sigma}^2_{\hat{\beta}_Z} \tag{3.3}$$

Where the Sobel test statistic (defined as $z_{\hat{\beta}_Z\hat{\beta}_r} = \frac{\hat{\beta}_Z\hat{\beta}_r}{\hat{\sigma}^2_{\hat{\beta}_Z\hat{\beta}_r}}$) is assumed to follow a standard Normal distribution (Sobel 1982). Whether the Sobel test statistic follows a standard Normal distribution or an asymmetric distribution is subject to controversy. We therefore also derive estimates of the standard error estimates via bootstrapping, and use 95% confidence intervals that account for such possible asymmetry (MacKinnon et al. 2004).

Our analysis here is hindered by two complications. The first concern is the appropriate spatial extent around treatment facilities to measure the effects of land use. The SDWP does not provide land use information, nor does it record the location of water sources used by facilities. The NHN Work Unit Limits provide boundaries for drainage basins - however, with an average area of 130,000 ha, these catchments are considerably larger than the buffer zones commonly employed in the literature. We also run models where land use characteristics are derived within a defined

radius around each plant, e.g., 5, or 10 kilometers. Secondly, it is important to identify spatial relationships between the variables in Equations (3.1) and (3.2), and failing to do so can lead to biased estimates and flawed inference. Such relationships arise due to shared environmental, geographical, or socioeconomic factors between neighbouring facilities (Pattanayak and Butry 2005). We use the Moran's I statistic to test for spatial auto-correlation in the residuals obtained from Model 1. Moran's I measures spatial dependence between locations by comparing the similarity of one observation to that of its neighbours. A high Moran's I indicates strong spatial auto-correlation, meaning that nearby treatment facilities face similar costs. However, the calculated test statistics for Moran's I depends on the criteria used to identify a neighbour, specifically the used distance threshold. Selecting a certain threshold determines whether an observation is considered adjacent to another, and can impact whether we find any evidence of spatial auto-correlation. In this study, the distance threshold was determined by estimating the Moran's I test statistic for a sequence of distances and selecting the distance that corresponded to the first local peak as the value that best captured the underlying spatial structure of the data[1].

Once we establish that spatial patterns exist, we must identify an appropriate method to address this auto-correlation. We test two approaches to modelling spatial relationships in the present study. Spatial lag and spatial error models explicitly account for the spatial dependencies by incorporating spatially weighted terms to capture the impact of an observation on its neighbours. A spatially lagged dependent variable is incorporated to capture potential spillover effects across neighbouring observation, while a spatial error term corrects for auto-correlation in the residuals that reflect the influence of unobserved parameters. Formally, the spatial model can be expressed as:

$$y = \rho_y W y + r\beta_r + Z\beta_Z^{'} + X\beta_x + \epsilon_y \tag{3.4}$$

$$\epsilon_y = \lambda_y W + u_y \quad E[u_y] = 0 \tag{3.5}$$

[1]This practice is recommended by the ArcGIS Pro documentation (Esri 2024).

$$r = \rho_r W r + Z\beta_Z + \epsilon_r \tag{3.6}$$

$$\epsilon_r = \lambda_r W + u_r \quad E[u_r] = 0 \tag{3.7}$$

Where y_i in Equation (3.4) represents treatment facility i, and ρ captures the influence of neighbouring facilities. W is the spatial weighting matrix. In Equation (3.5), λ captures the strength of the spatial error auto-correlation. In constructing the spatial weighting matrix, W, we apply the queen contiguity method, where observations are considered neighbours if they share a border. Adjacent observations are identified subject to a distance threshold and used to construct a neighbours list. Observations with no neighbours are assigned a weight vector of zero, indicating no direct spatial influence.

The inclusion of the spatial lag term changes our interpretation of estimated coefficients, as the dependent variables y and r now appears on both sides of Equations (3.4) and (3.6). Rearranging Equation (3.4), we can describe $S_{k,y}(W) = (I_n - \rho_y W)^{-1}(I_n\beta_k + Wy)$, where $S_k(W)_{ii}$ expresses the impact of a change in x_{ik} as a combination *direct* and *indirect* spatial effects (LeSage and Pace 2009). Here, k denotes a particular explanatory variable, such as land use or water quality. For example, an increase in an explanatory variable at facility i that impacts treatment costs at facility i can be denoted by $\frac{\partial y_i}{\partial x_{ik}} = S_{k,y}(W)_{ii}$. This is the direct effect. However, facility i's treatment costs are also determined by its neighbours. The *indirect effect* of an increase in an explanatory variable at a neighbour can be represented by $\frac{\partial y_i}{x_{ik}} = S_k(W)_{ij}$. The total spatial effect of land use on treatment costs is then represented by $S_{k,y}(W)_{ii} + S_{k,y}(W)_{ij}$.

Finally, `"lavaan"` in R allows for the bootstrapping of standard errors for the mediated effect of land use through turbidity. Bootstrapped standard errors are computed by repeatedly resampling the data with replacement and estimating the model on each resample. The standard errors of the mediated effect are then derived from the distribution of the estimates across these resamples. To incorporate bootstrapped standard errors in addition to spatial autocorrelation, we employ an ap-

proximation of an effective sample size using `"lavSpatialCorrect"` for a spatial correction to standard errors before proceeding with bootstrapping. Briefly, the effective sample size method adjusts for spatial autocorrelation by using the Moran's I to quantify the similarity between nearby observations, reducing the effective number of independent observations in the dataset to correct standard errors accordingly (see Appendix for details).

Thus, the study employs a rigorous methodological approach and accounts for various complexities in the relationship between land use, turbidity, and drinking water treatment costs. In accounting for shared variables, the spatial structure of our data, and the pathway through which land use influences drinking water treatment, we uncover the value of watersheds ecosystem services to water resource management in Canada. The following section will describe the findings of our research.

3.4 Data

The study makes use of Statistics Canada's 2015 Survey of Drinking Water Plants (SDWP), a biennial census of drinking water plants that are licensed and regulated by provincial agencies (excluding those serving fewer than 300 people, schools, commercial establishments, First Nations communities, etc.). Following 2015, the SDWP began collecting information on drinking water plants at the municipal level. As the purpose of this study is to explore the relationships between land cover the location of a facility, and how this impacts turbidity and drinking water treatment costs, we use the 2015 SDWP for this analysis. Table 3.2 presents summary statistics for the drinking water treatment facilities of Canada. The 2015 SDWP reports extensively on the costs of drinking water production. Cost data is reported annually. Costs are broken down into two categories: capital expenditures and operation and maintenance (O&M) costs. The former category includes capital costs for the acquisition and treatment of raw water, along with investment in physical assets including machinery and processing infrastructure. The latter category can be further broken down into materials, labour, energy, and other expenditures. O&M costs will be the

dependent variable in the present analysis. Costs associated with the distribution of drinking water are not included in this variable.

Table 3.2: Summary of Dependent and Independent Variables

Variable Type	Description	Obs.	Mean	St. dev.
Dependent variable				
Total O&M	Total operating and maintenance costs for treatment facility in 2015 ($CAD).	410	1,375,615	2,747,272
Facility characteristics				
Groundwater	Proportion of raw water intake from groundwater sources in 2015 (%).	410	1.38	7.82
Population Served	Population served by treatment facility in 2015	410	54,946.70	169,272.83
Days > 90	Number of days spent at over 90% capacity in 2015.	400	12.04	59.82
Residential	Proportion water produced for residential use, 2015 (%).	400	58.52	21.43
Losses	Proportion water produced that was lost in the distribution system, 2015 (%).	400	14.47	13.17
Wholesale	Proportion water produced that was sold to other municipalities, 2015 (%).	400	5.25	15.07
ICI	Proportion of water produced for industrial, commercial, or institutional use, 2015 (%).	400	21.55	15.29
Coagulation	Dummy variable, if coagulation technologies were used at facility in 2015.	410	0.73	0.44
Pretreatment	Dummy variable, if pretreatment technologies were used at facility in 2015.	400	0.56	0.50
Filtration	Dummy variable, if filtration technologies were used at facility in 2015.	410	0.84	0.37
Volume	Total volume produced in 2015 (m^3).	410	8,559,166	27,811,572
Turbidity	Maximum average monthly value of turbidity (NTU[2])	380	20.8	51.5
land use characteristics				

Continued on next page

[2]Turbidity is recorded in Nephelometric Turbidity Units, a measure of the concentration of suspended solids and organic matter in water (Price et al. 2017).

Table 3.2 continued from previous page

Variable Type	Description	Obs.	Mean	St. dev.
Forest	Proportion forest cover within sub-sub drainage area in 2015 (%).	410	44.97	28.50
Cropland	Proportion agriculture within sub-sub drainage area in 2015 (%).	410	19.04	24.37
Urban	Proportion urban cover within sub-sub drainage area in 2015 (%).	410	3.80	4.21
Wildfire	Proportion sub-sub drainage area experiencing forest fire in 2015 (%).	410	0.031	0.32
Labour	Median hourly wage rate of utility employees by province in 2015 ($CAD/hr).	380	39.52	2.12
Electricity	Highest electricity rate in province where facility is located in 2015 (¢/kWh).	380	5.67	1.06

The biennial census further includes a description of the raw water treatment processes employed by facilities in Canada. Such detail is rare: previous watershed ES valuation studies controlled for the differences between plants typically by using fixed effects models, e.g., with plant-level fixed effects (Vincent, et al. 2016; Gelo and Turpie 2022), or municipality characteristics (Abildtrup et al. 2013). Data on the use of various treatment processes were consolidated into four dummy variables indicating the use of pre-treatment, coagulation, disinfection, and filtration methods.

Three variables relating to raw water quality are recorded in the survey. *Escherichia coli* (E. coli), turbidity, and temperature are all reported monthly, with minimums, maximums, and averages for each parameter. Our raw water quality proxy is the mean of the maximum monthly turbidity recorded at a treatment facility in 2015, as it is often used as an indicator of other constituents of water quality (Crittenden et al. 2012). In order to reconcile time scales between treatment costs and water quality, the turbidity measure used in this study is the average maximum turbidity reported at each plant over the course of 2015. According to Vincent et al. (2016), a short-run cost function for a firm that uses one or more unpriced environmental inputs includes four types of variables: (1) the firm's output level, (2) prices paid by the firm for labour and other

non-environmental inputs, (3) the quantity of capital and other fixed factors used by the firm, and (4) the quantity of environmental inputs used. We use the population served by a treatment facility as a proxy for its total output. Factor prices are not included in the SDWP; the effects of prices paid by firms for inputs are captured following the approach of Price et al. (2017). The price of labour is taken as the median wage rate of utility employees by province in 2015, as reported by Statistics Canada's Table 14-10-0064-01 (Employee wages by industry, annual). Electricity rates paid by treatment facilities are represented by Hydro Quebec's *2015 Comparison of Electricity Prices in Major North American Cities*. Dummy variables indicating the use of pre-treatment, coagulation, and filtration are included as a rough proxy for quantity of capital and other fixed factors in Canadian treatment facilities. Finally, the quantity of environmental inputs is measured by land use proportions around treatment facilities.

A significant proportion of treatment facilities provide no response to questions regarding water quality. Many observations are dropped due to this. Facilities that predominantly use groundwater sources are overrepresented in these missing observations. Of those treatment facilities that remain, there are furthermore differences in the frequency of monitoring raw water quality. For example, in 2013, 42% of drinking water treatment facilities with surface water intake monitored continuously, 34% monitored daily, and the remaining 24% monitored with less frequency (Statistics Canada 2017).

Data from the 2015 SDWP is paired with three publicly available datasets from the Government of Canada. The 2015 Land Cover of Canada (LCC) is the source for all land cover information in the study. The LCC has a spatial resolution of 30m (with each pixel representing a 30m-by-30m area). Classification accuracy typically decreases with coarser resolution (Singh et al. 2013), and an assessment of the 2015 map found an accuracy of 79.90%. The LCC contains 19 land use categories, six of which correspond to forest types. Urban settlement and cropland record human activity. Remaining land use types (indicating shrubland, grassland, wetland, barren land, water, and snow and ice) are amalgamated into a natural land use category and are employed as the baseline.

Ideally, land use upstream of a treatment facility would be assessed by delineating the specific watershed that drains into a facility's intake point. However, the 2015 SDWP does not include data on precise intake locations. According to the 2003 Standard Drainage Area Classification (SDAC), Canada can be divided into 11 major drainage areas; other sources put this figure at 25 (Statistics Canada 2009). However, using the major drainage area boundaries would result in little between-facility variation in land use. Instead, we use the National Hydro Network (NHN), which provides geospatial data regarding Canada's inland surface waters. In the context of calculating land use proportions around a treatment facility without information on intake points, we utilise the NHN Work Unit Limits, which define the boundaries of Canada' sub-sub drainage areas. Canada's 11 major drainage areas can be subdivided into 974 sub-sub drainage areas; these ideally reflect local hydrological characteristics within larger basins, and they allow us to include greater variation in land cover around a facility. However, these boundaries cover large areas, with the average drainage basin covering 130,000 hectares in area. Alternative approaches to land cover analysis include, for example, estimating land use proportions in buffer zones of some radius around an intake location. However, it is unclear what the appropriate distance is for such a boundary. Mulatu et al. (2021) show that the contribution of forest cover to treatment cost reduction decreases as the buffer distances increases from 2.5 to 30 kilometers. While the NHN sub-sub drainage boundaries serve as the basis for our analysis, we also estimate models where land use proportions are derived for buffers of 5km and 10km around a treatment facility subject to the strong assumption that treatment facilities are generally located close to their water intake.

Finally, the Canada Landsat Disturbance (CanLaD) 2017 records forest areas affected by fire or harvest in Canada between 1984 and 2015. Changes are detected based on Landsat mosaics collected in the summers of 1984-2015. The infrequency of data collection leads to a high classification error - a fire occurring in October 2014 will be detected in July 2015 and assigned to the latter year. The CanLaD has a classification accuracy of 69% for year of forest disturbance, this increases to 99% if the following year is included in the classification criteria (Guindon et al. 2018). As with the 2015 LCC, the proportion of area lost to fire is estimated for each sub-sub

drainage area as per the NHN Work Unit Limits.

3.5 Results

3.5.1 Spatial Regression Models

Table 3.3 presents the results of the spatial lag and spatial error models, using the specification described by Equations (3.4)-(3.7). Columns (1), (3), and (5) describe coefficient estimates for the regressions on turbidity, where land use shares have been derived within 5km, 10km, and the sub-sub drainage basin of a facility, respectively. As anticipated, land use influences turbidity. However, considering proximity is crucial to understanding this relationship. Forest cover is associated with a statistically significant negative impact on turbidity relative to other natural land use at more localised scales, within 5km and 10km of a facility, although this effect diminishes in magnitude and is not significant at the sub-sub drainage scale. Cropland is associated with a statistically significant positive effect only at the sub-sub drainage scale, while urban cover is never associated with a statistically significant impact on turbidity. Further, wildfires within the sub-sub drainage area of a treatment facility have a large impact on turbidity. A wildfire with an extent of 1300ha (13 km^2, 1% the size of the average basin) is associated with a 122% increase in turbidity. This coefficient estimate is orders of magnitude higher than coefficients for other land use categories.

The spatial lag and error terms provide insight into the relationship between land use and turbidity at neighbouring facilities. The spatial lag term (ρ_r) is always positive, although it is only significant at the localised scales. With the spatial lag term, changes in land use at one facility influence turbidity not only locally, but also at neighbouring facilities due to spillover effects. Intuitively, statistical insignificance at the sub-sub drainage scale reflects the fact that NHN boundaries represent independent hydrological units. The spatial lag term loses significance because facilities within the same sub-sub drainage basin share the same land use characteristics, and neighbouring facilities in different drainage basins experience less spatial dependence. To

interpret the regression coefficients correctly, we also need to consider spatial indirect, and total effects. By considering these effects, we capture both the immediate and broader impacts of land use across connected areas. Table 3.6 in the Appendix contains direct, indirect and total *spatial effects* for the regression of land use on turbidity. Table 3.6 shows that, across all spatial scales, the indirect effect of each land use type on turbidity is larger than the direct effect. Land use changes surrounding a facility significantly impact turbidity levels at neighboring facilities through spatial spillovers. For instance, although forest cover has a modest effect on reducing turbidity locally, its indirect spatial effects, particularly at smaller scale, are much larger, so that forested areas around neighboring facilities substantially contribute to lower turbidity levels. This pattern emphasises the importance of considering both direct and indirect effects to fully understand the broader influence of land use on water quality. The spatial error term (λ_r) is also statistically significant at the 10km and sub-sub drainage scale, though its sign differs across these scales. While unobserved spatially correlated factors affect turbidity, the effect depends on the spatial extent considered.

Coefficient estimates for the regressions on variable drinking water treatment costs are found in Columns (2), (4), and (6) of Table 3.3, again with land use shares derived within 5km, 10km, and the sub-sub drainage area of a facility, respectively. As anticipated, higher turbidity is associated with an increase in treatment costs. The coefficient estimate for turbidity is interpreted as its cost elasticity, which in the model is estimated to be quite small. A 1% increase in turbidity increases costs in the range of 0.032%-0.043%. This is far lower than the 0.19% estimated by Warziniack et al. (2017) in the US, or the 0.26% by McDonald and Shemie (2014). However, the coefficient is comparable to Horn (2011)'s elasticity estimate of 0.03% for total organic carbon (TOC) in Japan, and Price et al. (2017)'s estimate of 0.099% in Canada using the 2011 SDWP. Surprisingly, the coefficient is not statistically significant at any spatial scale. However, at the 5km and sub-sub drainage scales, the coefficients have p-values of 0.1185 and 0.1255 respectively, indicating a near-significant association.

The regression on treatment costs additionally controls for treatment facility characteristics, treatment technologies, and input factors. The coefficient for population served is statistically

Table 3.3: Spatial Regression Results with Land Use Characteristics

	5km Buffer		10km Buffer		Sub-sub drainage	
Variables	(1) Turbidity	(2) Costs	(3) Turbidity	(4) Costs	(5) Turbidity	(6) Costs
Forest	-0.419**	-0.338*	-0.445***	-0.813***	-0.056	-0.626**
	(0.165)	(0.184)	(0.161)	(0.176)	(0.193)	(0.143)
Agriculture	0.305	0.130	-0.043	-0.298	1.691***	-0.424*
	(0.374)	(0.328)	(0.414)	(0.365)	(0.375)	(0.223)
Urban	0.401	1.410	0.661	1.499	-1.896	1.699
	(1.311)	(0.991)	(1.319)	(0.998)	(1.975)	(1.160)
Wildfire	-69.552	-475.860	-20.167	38.231	79.794***	13.335
	(651.021)	(442.260)	(65.458)	(39.009)	(14.050)	(16.102)
Turbidity		0.043		0.032		0.043
		(0.275)		(0.029)		(0.028)
Population		-0.492***		-0.510***		-0.484***
		(0.038)		(0.038)		(0.032)
Seasonal		0.071***		0.093***		0.060**
		(0.025)		(0.025)		(0.025)
Groundwater		0.001		0.002		0.003
		(0.004)		(0.004)		(0.004)
Residential		0.003*		0.004*		0.002
		(0.002)		(0.002)		(0.002)
Losses		0.003		0.005*		0.005
		(0.003)		(0.003)		(0.003)
ICI		-0.002		-0.002		-0.002
		(0.003)		(0.003)		(0.003)
Days $> 90\%$		0.058**		0.050*		0.048*
		(0.026)		(0.027)		(0.027)
Filtration		0.609***		0.796***		0.776***
		(0.129)		(0.132)		(0.129)
Coagulation		0.057		-0.036		0.133
		(0.105)		(0.068)		(0.109)
Pre-treatment		0.056		-0.036		4e-4
		(0.065)		(0.068)		(0.067)
Electricity		-0.328		0.480		0.167
		(0.390)		(0.244)		(0.263)
Labour		2.005***		1.838***		1.955***
		(0.185)		(0.125)		(0.131)
ρ	0.767***	-0.287***	0.775***	-0.021	0.095	-0.156*
	(0.043)	(0.043)	(0.042)	(0.081)	(0.170)	(0.090)
λ	-0.639	0.849***	-0.643***	0.661***	0.397***	0.835***
	(0.096)	(0.043)	(0.095)	(0.119)	(0.153)	(0.061)
Constant	0.600***	2.036***	0.660***	-0.152	1.669***	0.866
	(0.158)	(0.444)	(0.158)	(0.487)	(0.416)	(0.615)
No. observations	389	374	389	377	389	377
AIC	1362.3	855.0	1363.0	888.7	1341.0	873.8
AIC for OLS	1428.5	899.6	1431.5	896.8	1372.4	875.5
Pseudo-R^2	0.353	0.893	0.358	0.880	0.273	0.887

[1] Note: * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$.

significant and positive; a 1% increase in the population served by a facility correlates with a reduction of at least 0.484% in *per capita* treatment costs. Seasonal variation in volume treated (represented by the standard deviation of the monthly volume in 2015) is also significant. A 1% rise in variability correlates to an at least 0.060% increase in costs. Although the use of coagulation and pre-treatment technologies does not significantly impact treatment costs, the use of filtration increases costs by over 83.8%. Further, we find that the sector a facility serves significantly impacts costs. Facilities serving a higher proportion of residential end-users tend to have lower per capita treatment costs relative to wholesale distribution, with a positive, statistically significant effect. There is a statistically significant relationship between the number of days a firm spends at near-full capacity and costs, with a 1% increase in the number of days at over 90% capacity correlating to a 0.058% unit increase in treatment costs. Finally, there is a strong association between labour prices and treatment costs, but electricity prices show no significant relationship with operating expenses.

The spatial parameters again provide insights into the determinants of drinking water treatment costs. Contrary to expectations, we find no evidence of technical spillover between facilities. The spatial lag term (ρ_y) is negative, suggesting competitive rather than collaborative dynamics may drive treatment costs. Table 3.7 in the Appendix contains direct, indirect and total *spatial effects* for the regression on drinking water treatment costs. The spatial error term (λ_y) is also positive and significant at all spatial scales, indicating that the residuals are correlated across space.

Finally, the spatial regression model exhibits stronger predictive power than the OLS model, as reflected by consistently lower AIC values across all spatial scales. For turbidity, the AIC values remain relatively stable at the more localised scales, but drop at the largest scale. However, the opposite pattern is determined with respect to the R^2, as the models find explanatory power drops with increasing scope for land use shares. This may affect the reduced variation in land use characteristics for treatment facilities within the same drainage area, which reduces the model's capacity to explain turbidity. There is a more consistent pattern with respect to the regressions on treatment costs. Both the AIC and the pseudo-R^2 peak within the 5km scale, and incorporating

more information regarding land use results in reduced explanatory and predictive power.

3.5.2 Structural Equation Models

While we are interested in learning how spatial dependencies impact the relationship between land use, turbidity, and drinking water treatment costs, we also wish to carry out inference on the mediated effect of land use on treatment costs. Table 3.4 presents the regression results for the SEM described by Equations (3.1) and (3.2), where standard estimates for coefficients have been corrected for spatial autocorrelation using the effective sample size method. Coefficient estimates for the impacts of land use on turbidity are again in Columns (1), (3), and (5). Forest cover is again associated with a statistically significant negative effect on turbidity relative to other natural land use. Cropland and urban cover are associated with a strong positive influence on turbidity, particularly at more localised scales. Surprisingly, the effect of wildfire disturbance on turbidity is less clear, with no statistically significant coefficient at the 10km or sub-sub drainage scales (the variable is omitted in the 5km regression due to a lack of non-zero observations). Further, while the coefficients for land use are greater in magnitude using the SEM specification than those in Table 3.3, note that they are comparable to estimates of the total spatial effect (see Table 3.6 in the Appendix), suggesting that the SEM specification reasonably captures the effects of land use on turbidity.

Columns (2), (4), and (6) present the regressions on drinking water treatment costs. The results are broadly consistent with the findings of the spatial lag and spatial error model in Table 3.3. The coefficient for turbidity maintains its magnitude, and is statistically significant at all spatial scales. The sign and magnitude of the population served, seasonality, filtration, and labour prices additionally remain unchanged. End-use and days spent at over 90% capacity are no longer associated with a significant effect on treatment costs. The coefficient for forest cover in the regression on treatment costs is statistically significant and negative at every spatial scale, suggesting a strong residual impact on treatment costs relative other natural land use. However, hypothesis testing via the Wald test finds that the difference between forest cover, urban cover, and cropland is not statis-

tically significant at any spatial scale. In contrast, wildfires that occur within 10km of a treatment facility have a positive, statistically significant residual impact on treatment costs, but this is not true at the sub-sub drainage scale. While it is unexpected that we do not find a significant effect on turbidity in the aftermath of a fire disturbance, it is reasonable that wildfires closer to a facility impact treatment costs directly. Wildfires can be associated with other costs, for example, through damage to pipes and infrastructure near the site of a disturbance that need repair and replacement (Smith et al. 2011).

The adjusted R^2 of the SEM specification also reflects the findings of the spatial lag and spatial error models. The explanatory power of the regression on turbidity initially increases with the scope of the land use variables, and drops in the final model. The explanatory power of the regressions on treatment costs is stable across spatial scales, indicating that the specification explains 66.3% to 67.4% of the variation in the data.

Table 3.5 presents inference on the mediated effect. Statistical significance, determined using the Sobel test statistic as described by Equation (3.3), where Columns (4) and (5) present the coefficients and standard errors of the mediated effect, respectively. The coefficient for the mediated effect $\beta_Z\beta_r$, can also be interpreted as the percentage change in treatment costs implied by a 1% change in land cover through the influence land use has on turbidity, Column (2) describes the land use change implied by each spatial scale in hectares. Additionally, Column (6) presents the asymmetric 95% confidence intervals for the mediated effect using bootstrapped standard error estimates, as recommended in the mediation analysis literature (MacKinnon and Webb 2017). Column (7) provides monetary estimates of the impact of land use surrounding a treatment facility has on its costs. To convert coefficient estimates into monetary terms, we leverage the fact that the average in-sample treatment facility has annual costs of $1,375,615. For example, a 1% increase in forest cover at the expense of other natural land use within a 5km distance of a treatment facility is associated with a change in turbidity of -1.216%, while a 1% change in turbidity is associated with a change in treatment costs of 0.046%. The product of these two terms is the mediated effect, in this case implying a reduction in treatment costs by 0.056%, or savings of roughly $770 generated

from converting 78 hectares of other natural land use to forest. This translates into per hectare savings of $9.81 CAD/ha/yr. Similarly, 1% increases in cropland and urban cover are associated with cost increases of approximately $715 and $907, respectively. These per hectare costs and savings decrease as the spatial scale increases, where a 1% change in land use represents a larger area, but the influence of distant land on treatment costs is comparatively smaller. Within 10km and at the sub-sub drainage scale, conversion of 1% of the land from other natural land use to forestland is associated with savings of $1.93 CAD/ha/yr and $0.39 CAD/ha/yr (although only the latter term is statistically significant).

While not all coefficients are statistically significant relative to natural land use in Table 3.5, we use the Wald test to identify when the differences between forest cover and urban cover or cropland are statistically significant. At the 5km scale, the coefficient for forest cover is significant relative to both agriculture ($\chi^2 = 4.502, p = 0.034$) and urban cover ($\chi^2 = 4.921, p = 0.027$). A 1% increase in forest cover at the expense of cropland within a 5km distance of a treatment facility is associated with a 0.11% decrease in treatment costs (the difference between the coefficients for the mediated effects), or savings of $1,485 generated from converting 78 hectares of cropland to forest. This translates into per hectare savings of $18.92 CAD/ha/yr. Relative to urban cover, the figure is $21.37 CAD/ha/yr. The differences are also significant at the 10km scale ($\chi^2 = 4.444, p = 0.034$ for agriculture, $\chi^2 = 5.378, p = 0.020$ for urban cover), although the per hectare savings again decline with distance. Within 10km, conversion of 1% of the land from cropland or urban cover to forestland is associated with savings of $3.81 to $5.78 CAD/ha/yr, respectively. At the sub-sub drainage scale, the difference is only significant relative to agriculture ($\chi^2 = 3.828, p = 0.050$), although we do not find a statistically significant difference between forest and urban cover ($\chi^2 = 1.539, p = 0.215$). At the sub-sub drainage level, savings are estimated at $1.28 CAD/ha/yr relative to agriculture. These results underscore the cost-saving potential of forest conservation near treatment facilities, with the strongest benefits observed within 5km, where land use changes yield the highest per hectare savings in treatment costs.

Table 3.4: Structural Equation Model Regression Results

	5km Buffer		10km Buffer		Sub-sub drainage	
Variables	(1) Turbidity	(2) Costs	(3) Turbidity	(4) Costs	(5) Turbidity	(6) Costs
Forest	-1.216***	-0.624*	-0.905**	-0.920**	-0.870***	-0.394**
	(0.431)	(0.327)	(0.454)	(0.362)	(0.314)	(0.173)
Agriculture	1.116**	-0.401	0.884	-0.710	1.982***	-0.235
	(0.522)	(0.402)	(0.549)	(0.446)	(0.390)	(0.203)
Urban	1.431***	-0.540*	1.811***	-0.804**	-3.510*	0.985
	(0.442)	(0.294)	(0.421)	(0.327)	(1.843)	(1.172)
Wildfire	-	-	-0.043	27.703**	-12.453	58.352
	-	-	(23.783)	(13.255)	(56.159)	(45.931)
Turbidity		0.046**		0.048**		0.042*
		(0.022)		(0.022)		(0.022)
Population		-0.487***		-0.480***		-0.495***
		(0.047)		(0.043)		(0.039)
Seasonal		0.054**		0.053**		0.056**
		(0.025)		(0.024)		(0.024)
Groundwater		0.002		0.002		0.002
		(0.004)		(0.004)		(0.004)
Residential		-0.000		0.000		0.000
		(0.002)		(0.002)		(0.002)
Losses		0.001		0.001		0.001
		(0.003)		(0.003)		(0.003)
ICI		0.001		0.001		0.000
		(0.004)		(0.004)		(0.004)
Days $> 90\%$		0.044		0.040		0.046
		(0.035)		(0.036)		(0.034)
Filtration		0.690***		0.696***		0.682***
		(0.183)		(0.178)		(0.175)
Coagulation		0.147		0.164		0.148
		(0.113)		(0.113)		(0.106)
Pre-treatment		-0.004		-0.006		-0.010
		(0.077)		(0.078)		(0.073)
Electricity		0.074		0.090		0.027
		(0.353)		(0.314)		(0.034)
Labour		1.734*		1.707**		0.054***
		(0.749)		(0.708)		(0.018)
Constant	1.833***	1.270	1.701***	1.468	1.520***	5.230***
	(0.299)	(3.280)	(0.313)	(3.022)	(0.110)	(0.988)
No. observations	406	377	406	377	406	377
Adjusted R^2	0.211	0.663	0.227	0.673	0.175	0.674
F-statistic	47.51		46.43		66.91	

[1] Note: * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$.

Table 3.5: Inference on the Mediated Effect for Land Use

(1) Spatial Scale	(2) 1% change (ha)	(3) Variable	(4) $\beta_{\mathbf{Z}}\beta_r$	(5) $\sigma_{\hat{\sigma}_{\hat{\beta}_{\mathbf{Z}}\hat{\beta}_r}}$	(6) 95% CI	(7) $CAD/ha/yr
5km Buffer	78.540	Forest	-0.056*	0.032	-0.124 – -0.005	-9.81
		Agriculture	0.052	0.036	0.001 – 0.134	9.11
		Urban	0.066*	0.038	0.007 – 0.148	11.56
		Wildfire	-	-	-	-
10km Buffer	314.159	Forest	-0.044	0.028	-0.105 – 0.000	-1.93
		Agriculture	0.043	0.035	-0.005 – 0.124	1.88
		Urban	0.088*	0.047	0.013 – 0.183	3.85
		Wildfire	-0.002	1.153	-2.330 – 2.496	-8.76
Sub-sub Drainage	1300	Forest	-0.037*	0.023	-0.083 – -0.001	-0.39
		Agriculture	0.084*	0.047	0.003 – 0.185	0.89
		Urban	-0.148	0.112	-0.405 – 0.010	1.57
		Wildfire	-0.525	2.384	-5.821 – 4.379	-5.54

[1] Note: * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$.

3.6 Conclusion

This paper contributes to the growing literature on ecosystem services by quantifying the water filtration benefits of forests relative to other land uses in the context of Canadian drinking water treatment facilities. Our analysis demonstrates that forest cover significantly reduces treatment costs through its impact on raw water quality. Several features distinguish this study from previous research endeavours. This is the first study to carry out a watershed ES valuation exercise at national scale in Canada coupling data from drinking water treatment facilities to surrounding land use and assessing the impact of the latter on raw water quality and treatment costs. Further, this is the first ecosystem service valuation to our knowledge that models the impact of wildfire disturbance on drinking water treatment costs. Wildfires in Canada are anticipated to increase in the coming decade, but their impact on treatment costs has not been quantified before.

A key finding of this study is the importance of spatial scale when considering the impacts land use have on drinking water treatment costs. We find that the advantages of natural land-cover for drinking water treatment facilities are most pronounced at very localised scales, as we obtain our highest per hectare estimates when using land use proportions within 5km buffer distances. Conversion of agriculture to forestland within a 5km buffer zone of a treatment facility generates

$18.92 CAD/ha/yr in savings. Relative to urban cover, forests save $21.37 CAD/ha/yr. While the magnitude of the effect does not increase with scale, we see that the impact is persistent over larger spatial scales, particularly for agriculture. At the sub-sub drainage scale, we find that converting agriculture to forestland is still associated with a statistically significant increase in turbidity, although the estimated treatment cost savings generated are low, $1.28 CAD/ha/yr.

These monetary estimates may be low relative to existing estimates found in the literature. While we find a very strong relationship between land use and turbidity, we estimate that treatment costs are rather inelastic to raw water quality. A 1% increase in turbidity is associated with an approximately 0.03-0.05% increase in treatment costs, depending on the model employed. A possible caveat to this result is the considerable number of missing observations in the database. The majority of facilities omitted from this study did not record turbidity in 2015. Whether groundwater-intake or other surface-fed treatment facilities could be more greatly impacted by turbidity may lead to higher estimates for the ecosystem services provided by forests.

We include data on land disturbance in Canada to test the hypothesis that wildfires can have large impacts on turbidity. We find evidence of the impact of wildfires on turbidity only after controlling for spatial dependencies. In the spatial lag and spatial error model at the sub-sub drainage scale, a wildfire covering 1% of a basin ($\sim$ 1300ha) is associated with a 122% increase in turbidity as a treatment facility within the basin. We also find evidence of a residual effect, where wildfires within 10km of a treatment facility can influence treatment costs through other pathways than turbidity. Again, these results highlight the necessity of considering spatial scale when determining the impact of land use on drinking water treatment costs. However, there is a data mismatch that impacts our ability to estimate the impact of wildfires on treatment costs. The 2017 CanLaD has a classification accuracy of 69% for forest disturbance, as wildfires in 2014 and 2016 have a chance of being assigned to 2015. Thus, our wildfire data is imperfect. Nevertheless, our study underscores the substantial economic value of the services provided by forests in supporting water treatment processes, and represents a significant step in understanding the impacts of wildfires on forest ES provision. By demonstrating the benefits of forest conservation and management, we

contribute to a more nuanced understanding of the interplay between land use, water quality, and treatment costs, offering valuable insights for policymakers and stakeholders interested in source water protection.

3.7 Appendix

3.7.1 Adjustment of standard errors by effective sample size

"`lavSpatialCorrect`" accounts for spatial auto-correlation using the effective sample size method, based on the understanding that non-independence of observations provides less than n units of information for a dataset of n observations (Dale and Fortin 2014).

Given a SEM model, obtain the residuals $\mathbf{e}$ for the endogenous variables:

$$\mathbf{e} = \mathbf{y} - \hat{\mathbf{y}}$$

where $\mathbf{y}$ is the observed data and $\hat{\mathbf{y}}$ is the fitted values from the model.

Next, construct a distance matrix $\mathbf{D}$ using the coordinates (x_i, y_i):

$$D_{ij} = \sqrt{(x_i - x_j)^2 + (y_i - y_j)^2}$$

Invert the distance matrix to create a weight matrix $\mathbf{W}$, setting diagonal elements to zero:

$$W_{ij} = \begin{cases} \frac{1}{D_{ij}}, & \text{if } i \neq j \\ 0, & \text{if } i = j \end{cases}$$

Calculate Moran's I for each residual vector $\mathbf{r}_k$:

$$I = \frac{n}{W} \frac{\sum_{i=1}^{n} \sum_{j=1}^{n} w_{ij}(u_i - \bar{u})(u_j - \bar{u})}{\sum_{i=1}^{n}(u_i - \bar{u})^2}$$

where n is the number of observations and $\bar{u}$ is the mean of the residuals u_i. w_{ij} is the spatial weight between observations i and j, while W is the sum of all spatial weights.

The effective sample size, n_{eff}, then is used to adjust standard errors based on the significance

of Moran's I:

$$n_{\text{eff}} = \begin{cases} n, & \text{if } p_{\text{value}} > \alpha \\ n \cdot \frac{1-I}{1+I}, & \text{if } p_{\text{value}} \leq \alpha \end{cases}$$

Adjust the variance-covariance matrix $\mathbf{V}$ using the effective sample size:

$$\mathbf{V}_{\text{adj}} = \mathbf{V} \cdot \frac{n}{n_{\text{eff}}}$$

For each parameter θ_j in the model, calculate the adjusted standard error:

$$\text{SE}_{\text{adj}}(\theta_j) = \sqrt{\mathbf{V}_{\text{adj},jj}}$$

3.7.2 Spatial Effects

Table 3.6: Direct, Indirect, and Total Spatial Effects of Land Use on Turbidity at 5km, 10km, and Sub-Sub Drainage Area

Variable	**5km**			**10km**			**Sub-Sub DA**		
	Direct	**Indirect**	**Total**	**Direct**	**Indirect**	**Total**	**Direct**	**Indirect**	**Total**
Forest	-0.486	-1.315	-1.801	-0.519	-1.456	-1.975	-0.056	-0.006	-0.062
Agriculture	0.354	0.958	1.312	-0.051	-0.143	-0.194	1.693	0.176	1.869
Urban	0.465	1.258	1.724	0.771	2.163	2.934	-1.898	-0.198	-2.096
Wildfire	-80.682	-218.237	-298.919	-23.536	-65.992	-89.528	79.878	8.321	88.198

Table 3.7: Direct, Indirect, and Total Spatial of Determinants of Drinking Water Treatment Cost at 5km, 10km, and Sub-Sub Drainage Area

Variable	**5km**			**10km**			**Sub-Sub DA**		
	Direct	**Indirect**	**Total**	**Direct**	**Indirect**	**Total**	**Direct**	**Indirect**	**Total**
Turbidity	0.043	-0.010	0.033	0.032	-6.60e-4	0.031	0.043	-0.006	0.037
Population	-0.493	0.111	-0.382	-0.510	0.011	-0.499	-0.484	0.066	-0.418
Seasonal	0.071	-0.016	0.055	0.093	-0.002	0.091	0.060	-0.008	0.052
Residential	0.004	-7.94e-4	0.003	0.004	-8.01e-5	0.004	0.002	-3.20e-4	0.002
Losses	0.004	-7.89e-4	0.003	0.005	-1.04e-4	0.005	0.005	-6.17e-4	0.004
ICI	-0.002	5.48e-4	-0.002	-0.002	4.50e-5	-0.001	-0.002	2.90e-4	-0.002
Days > 90	0.058	-0.013	0.045	0.050	-1.04e-3	0.049	0.048	-0.006	0.041
Filtration	0.511	-0.138	0.473	0.796	-0.017	0.779	0.776	-0.105	0.671
Coagulation	0.057	-0.013	0.044	0.106	-0.002	0.104	0.133	-0.018	0.115
Pretreatment	0.057	-0.013	0.044	-0.036	7.42e-4	-0.035	4.49e-4	-6.09e-5	3.88e-4
Electricity	-0.330	0.074	-0.256	0.480	-0.010	0.470	0.167	-0.023	0.145
Labour	2.011	-0.453	1.558	1.838	-0.004	1.800	1.956	-0.265	1.691
Forest	-0.339	0.076	-0.263	-0.812	0.017	-0.796	-0.626	0.085	-0.541
Agriculture	0.131	-0.029	1.011	-0.298	0.006	-0.292	-0.525	0.057	-0.367
Urban	1.414	-0.319	1.096	1.499	-0.031	1.468	1.700	-0.230	1.470
Wildfire	-0.048	0.011	-0.037	38.231	-7.931	37.348	13.342	-1.807	11.534

Chapter 4

Contracting Under Uncertainty in Payments for Ecosystem Services

4.1 Introduction

Market-based instruments, such as Payments for Ecosystem Services (PES), have become an increasingly important tool for environmental conservation (Wunder 2015). Environmental economists envision the conservation problem as one of market failure. Ecosystem services (ES) like flood protection or water quality regulation are public goods and thus non-excludable, meaning once provided, they are available for anyone to use. Consequently, while their management and conservation provides end-users with benefits, it is difficult to charge a fee for their use (Carlson et al. 2015). Non-excludability makes it challenging to establish markets directly for the services ecosystems provide. Where successful, the marketisation of ES induces landowners to conserve ecosystems instead of converting land to profitable alternatives like agriculture.

PES schemes, where a "*well-defined ES is being bought by an ES buyer from an ES provider if and only if the ES provider secures ES provision*" (Wunder 2005, p.1) are the preferred instrument for marketisation. PES schemes can reduce ecosystem degradation, although their design

and implementation comes with significant challenges[1]. Many of these challenges relate to *conditionality*; for a scheme to be successful, payments to providers must be conditional on the supply of services. Payments to landowners are typically contingent on previously agreed-upon land-management practices, implicitly assuming that these actions will lead to some desired environmental outcome, and that the investment will be socially worthwhile, even when the mechanisms linking land use and ecosystem services are unclear (Porras et al. 2008; de Lima et al. 2017). In practice, it is unclear how often conditionality is actually achieved. Worryingly, many PES schemes do not monitor ES output, and the limited empirical evidence is concerning, suggesting that many PES schemes fail to achieve their environmental objectives (e.g., Brouwer et al. 2011; Börner et al. 2017; European Court of Auditors 2020; Wunder et al. 2020).

Ensuring conditionality is complicated by unpredictability. For example, there is much work in the PES literature on the impacts of *behavioural risk* arising from asymmetric information, where landowners have private knowledge on the costs of their actions. In the absence of compliance monitoring, landowners can leverage this asymmetry to their advantage, e.g., by accepting compensation for ES provision while foregoing contract-stipulated conservation efforts ("shirking"). In economics, the interactions between a principal (e.g., a municipality in the PES context) and an agent (landowner) are driven by the incentives created by a given contract structure, and the field that studies how a principal creates contracts to achieve efficient outcomes in the presence of asymmetric information is known as contract theory (Salanié 2003). The performance of PES schemes is also impacted by *environmental risk* that arises due to the natural variability of ES processes, i.e., randomness (Walker et al. 2003). However, this problem can be mitigated using insights from contract theory literature. While variables like precipitation and temperature are unpredictable, they can be understood as stochastic processes and incorporated into risk management strategies.

Various researchers have used the principal-agent framework to study the design of incentive-compatible contracts in agri-environmental contracts with asymmetric information (Choe and Fraser 1999; Moxey et al. 1999; Ozanne et al. 2001; Fraser 2002; Ozanne and White 2007), and environ-

[1] See Muradian et al. (2010) on the false dichotomy proponents sometimes present of PES as a win-win solution to conservation problems.

mental risk (Derissen and Quaas 2013). What is missing in this literature is a study of contract design under uncertainty. *Uncertainty* is distinct from *risk*[2] (Gorissen et al. 2015), in that it involves "*the lack of confidence that a decision-maker has about possible outcomes and/or the probabilities of these outcomes*" (Hamel and Bryant 2017, p. 2). Uncertainty is of immediate relevance to PES. Often, PES schemes are created even as the exact ES production function remains unknown (Zabel and Roe 2009; Wunder 2013). This arises from factors such as data limitations, estimation error, or incomplete knowledge of the mechanisms driving ES processes (Onatski and Williams 2003; Moallemi et al. 2020). As a result, a principal must contract landowners with limited knowledge about how specific land use and land-management practices will translate into the provision of ES. Where uncertainty is large (as is particularly the case for watershed services), there is concern that, "*the less realistic the scientific basis of a PES scheme, the more exposed it is to the risk of buyers questioning its rationale and abandoning payments*" (Wunder 2005, p. 3).

The central question we seek to address is: Given persistent uncertainty, how can we design PES contracts to ensure robust and outcomes that align with social welfare objectives? To answer this question, we employ robust optimisation techniques, which offer a valuable framework for tackling contract design under uncertainty. These methods have been explored in various economic contexts, particularly in agency theory (Carroll 2015; Carroll and Meng 2016; Yu and Kong 2020). The general approach involves a principal that lacks precise knowledge of the parameters governing an output distribution, but knows these parameters fall within some range, known as an uncertainty set. A principal can shield themselves from uncertainty by judging contracts based on their worst-case performance within the range of possibilities defined by the uncertainty set, i.e., by having max-min preferences. Such contracts offer a performance guarantee, ensuring that even under the least favorable conditions, the principal achieves a positive outcome. Robust techniques have been applied in environmental economics, particularly in the context of climate change. Athanassoglou and Xepapadeas (2012), Xepapadeas and Yannacopoulos (2017), and

[2]The two terms are often used interchangeably to refer to unpredictability. What we refer to as uncertainty has also been referred to as *model* or *parameter uncertainty* (Yu and Kong 2020), or *Knightian uncertainty* (Athanassoglou and Xepapadeas 2012) in the literature.

Rudik (2020) all utilise max-min robust control to address uncertainty in climate-related policy.

The novelty of this study is its application of robust optimisation to the design of PES contracts, addressing uncertainty in ES provision - a context where such methods have not been explored. We address two questions. Firstly, how can principals design robust PES contracts to manage uncertainty in the relationship between land management practices and ES provision? Secondly, how do adverse selection and moral hazard influence the optimal design of PES contracts in an uncertain environment? We start with the standard PES contract with and without adverse selection, assuming that the principal contracts with landowners on one verifiable input, land, and employs input-based payments. We then explore how introducing uncertainty into this setting impacts the contract offered by an uncertainty-averse principal and landowner. In this setting, the principal and landowner have limited knowledge of the potential of conservation to product ES. We show that with and without adverse selection, uncertainty-aversion motivates the principal to adopt a conservative approach to investing in ES, i.e., by reducing contracted land input, as this is the only way to secure a performance guarantee in the worst-case scenario with regards to ES provision. The landowner is not impacted by uncertainty surrounding the production of ES.

This finding is validated with a numerical exercise utilising data on soybean and grain corn farmers in Ontario in conjunction with the findings of the watershed ES valuation exercise in Chapter 3 on the benefits provided by forests to drinking water treatment facilities in Canada. We are further interested in how such contracts can be expected to perform relative to the standard contract where a principal does not care about uncertainty. Using a simple simulation, we show that there is a trade-off between robustness and contract performance. As the size of the uncertainty set increases, because worst-case outcomes have a small likelihood of occurring relative to other outcomes, the principal's conservative approach results in lower expected social welfare when uncertainty is realised and the probability distribution is known.

Finally, we explore two settings in the moral hazard framework. Many PES schemes are input-based, where payments are tied to land management practices rather than outcomes. However, output-based schemes are increasingly being discussed as a way to improve conditionality and ad-

dress moral hazard by linking payments to actual ES provision (Moxey and White 2014; Schwarz et al. 2008; Herzon et al. 2018). Our model reflects these two settings: the more common input-based contracts, and output-based contracts, which may produce better incentives for ES provision, but have not been studied in detail. In the first setting, the principal incentivises landowners to allocate land for conservation while effort is costly to observe. This design reflects real-world PES schemes that often rely on observable land use practices although effort and output are difficult to monitor. Our findings again show that a conservative approach to investing in ES is the best way to ensure a performance guarantee. In the second setting, payments are output-based, linking compensation to measurable ecosystem service outcomes. Here, the optimal scheme requires the principal to vary payments with output to induce effort by the landowner. With uncertainty, a hybrid contract is optimal. This contract includes a fixed payment to ensure participation. Further, a bonus payment structure where landowners receive a fixed share of the value of ES output, is sufficient to induce compliance. The impact on land input depends on the properties of the bonus payment structure and the costs of compliance, although we show that, as the cost of exerting effort increases, the principal is again likely to reduce investment into conservation.

The remainder of this paper is organised as follows. Section 4.2 provides a review of the existing literature, followed in Section 4.3 by a description of the standard model. In Section 4.4, we introduce the robust optimisation framework, describing our findings with and without adverse selection, followed by an illustration of our results with a numerical example. In Section 4.5, we explore the consequences of moral hazard on optimal contracts, followed by a brief policy discussion on the implications of our findings in Section 4.6. Finally, in Section 4.7, we conclude.

4.2 Literature Review

Grossman and Hart (1983) provide one of the earliest descriptions of the principal-agent problem in the context of employment and labour contracts. Early models typically focused on wages (e.g., Mirrlees 1976; Shavell 1979), insurance (Holmström 1979), and taxation (Guesneri and

Laffont 1978). Only more recently has the principal-agent problem been extended into the agri-environmental context, particularly in contracting landowners for ES provision (Ferraro 2008). Principal-agent models tackle two problems that arise from asymmetric information. Adverse selection (also known as hidden information) occurs when agents that have private knowledge of their type are able to extract an information rent from an unknowing principal. Moral hazard in the agri-environmental context typically refers to costly state verification (Ozanne and White 2007), where principals face challenges verifying the actions taken by landowners, leading to potential shirking. Both problems are exacerbated by the costliness of compliance monitoring and uncertainty regarding the verifiability of effort and ES output.

Our work contributes to the literature on optimal contracts in PES by incorporating uncertainty, where principals have limited distributional information on ES production. Our research is most closely related to the work of Choe and Fraser (1999), Moxey et al. (1999), Ozanne et al. (2001), Fraser (2002), Crepin (2003), Ozanne and White (2007), and Derissen and Quaas (2013). This literature has predominantly focused on behavioural risk. Adverse selection is modelled by assuming that landowners have private information about the costs of providing ES (Choe and Fraser 1999). Optimal contracts with adverse selection involve a rent-efficiency trade-off, with low-cost landowners receiving higher payments and high-cost landowners being contracted for less input (Moxey et al. 1999). With moral hazard, the focus shifts towards the actions landowners take after a contract is signed - actions that the principal cannot observe directly (Fraser 2002). Moral hazard is typically modelled by introducing monitoring and penalties to disincentivise non-compliance. Due to the cost of monitoring, there is again a reduction in contracted input (Ozanne et al. 2001). Derissen and Quaas (2013) differ from the remainder of the literature in two aspects. Firstly, they are the only study in this literature to consider environmental risk and a risk-averse principal. Secondly, while earlier studies generally only consider input-based payments[3], they show that output-based payments characterise optimal contracts in some settings with environmental risk.

Yet in each case, it is assumed that ES are well-defined, well-understood, and can be easily ex-

[3]Although there are exceptions, e.g., White and Hanley (2016) explore the consequences of a per-unit-of output and a lump-sum payment.

pressed in monetary terms - that there is no uncertainty. Uncertainty, which arises from limited information on ES due to the complexity of interactions between humans and ecosystems (Moallemi et al. 2020; Hou et al. 2013), poses significant challenges for PES. Many PES schemes are set up in environments where the relationships between land use and ES are uncertain (Muradian et al. 2010; Ojea 2015), which impacts their performance. For watershed services, PES schemes are often promoted on the basis of a precautionary strategy, rather than as a result of observed changes in service flows (Wunder and Alban 2008; Wertz-Kanounnikoff et al. 2011). The gap in the literature, therefore, lies in the insufficient integration of uncertainty regarding ES into research on the design of PES contracts. Only Li et al. (2023) consider this form of uncertainty. However, their goal is not to study optimal contracts in a setting with uncertainty, but rather to describe the performance guarantees of simple contracts that do not require prior distributional knowledge on the part of the principal. Moreover, this study explores uncertainty regarding the production of ES, while their focus is principal uncertainty regarding landowner costs of conservation.

There has been much work on extending the principal-agent framework to account for uncertainty by introducing robust optimisation techniques, particularly with reference to wage contracting (e.g., Carroll 2015; Carroll and Meng 2016). The robust optimisation problem is one where principals seek a performance guarantee by designing contracts to maximise their worst-case payoff. Ben-Tal and Nemirovski (1999) show how, given a set of assumptions about the nature of uncertainty, such problems can be expressed as convex optimisation problems. Yu and Kong (2020) apply this method to the agency problem with moral hazard for wages. Their framework does not assume a fixed probability distribution for outcomes but instead considers a range of possible scenarios, each of which can occur with unknown likelihood. By focusing on worst-case scenarios, this approach seeks to design contracts that are resilient to uncertainty. However, the general purpose of this literature is to show why certain contracts are so prevalent in practice, not to identify insights into managing uncertainty.

4.3 Expected Utility Analysis

4.3.1 First-Best Allocation

The principal (e.g., government or conservation authority) is a social welfare maximiser with the following objective function with respect to a target population of landowners, of which there are two types (indexed by $i \in \{l, h\}$):

$$\max_{x_l, x_h, F_l, F_h} \mathbb{E}_\phi[\sum_{i \in \{l,h\}} v(x_i) + (F_i - c_i(x_i)) - (1+\lambda)F_i] \tag{4.1}$$

The first term in Equation (4.1), $v(x_i)$, represents ES production, indicating the economic returns from conservation as a function of contract-stipulated input set by the principal, x, with $x \geq 0$. For simplicity, it is assumed that the only input is the amount (e.g., acres) of land set aside for ecological restoration, and that verifying landowner conservation is costless. $v(x_i)$ is stochastic with the following discrete probability distribution:

$$v(x) = \begin{cases} \phi_n x & \text{w.p.} a_n \\ \phi_{n-1} x & \text{w.p.} a_{n-1} \\ \vdots & \\ \phi_1 x & \text{w.p.} a_1 \\ \phi_0 x & \text{w.p.} 1 - \sum_{j=1}^{n} a_j \end{cases} \tag{4.2}$$

$\phi_j, j = 0, 1, ..., n$ represent the possible returns per unit of land set aside for conservation (marginal productivity), with $\phi_n > \phi_{n-1} > ... > \phi_1 > \phi_0$. Here, ϕ_0 represents the baseline marginal productivity of conservation. $a_j, j = 0, 1, ..., n$ are the probabilities associated with each return level $\phi_j x$, where $a_0 = 1 - \sum_{j=1}^{n} a_j$. The second term, $(F_i - c_i(x_i))$ represents the net payoff to landowner i from participation. Landowners are assumed to be risk-neutral. F_i is the compensation provided to the landowner with type-i, modelled here as a lump-sum payment. $c_i(x_i)$ is the conservation or compliance cost for a type-i landowner:

$$c_i(x_i) = \pi_i(x_i^*) - \pi_i(x_i^* - x_i) \tag{4.3}$$

The cost of ES provision is assumed to be monotonically increasing for all landowners, with $c_i^{'}(x), c_i^{''}(x) > 0 \forall x$. Equation (4.3) captures the opportunity cost of conservation for a landowner. $\pi_i(x)$ is the agricultural profit function. $\pi_i(x_i^*)$ denotes landowner profits when x_i^* is the profit-maximising input allocated to productive agricultural use. In turn, $\pi_i(x_i^* - x_i)$ corresponds to agricultural profit when x units of land are set aside for conservation. Spatial heterogeneity in the landowner population is also captured in the cost function. Type-h landowners are assumed to have higher-quality agricultural land than type-l landowners, and thus lose more from participating in a scheme, so that $c_h(x) > c_l(x) \forall x$.

The final term in Equation (4.1) considers the shadow cost of raising public funds to run a PES scheme. Public spending based on revenue generation by the principal to finance conservation results in a deadweight loss, with $\lambda \in (0, 1)$. Neither the conservation costs nor the deadweight loss generated from raising public funds are assumed to be impacted by environmental risk. In the absence of moral hazard and adverse selection, the principal's objective function in Equation (4.1) must be maximised subject to the following individual rationality (IR_i) constraints:

$$F_i - c_i(x_i) \geq 0 \forall i \in \{l, h\} \tag{4.4}$$

The IR_i constraints indicate that for a type-i landowner to participate in a PES scheme, the payment they receive must at least exceed their conservation costs. The absence of private information means that the principal is able to provide a take-it-or-leave-it contract to each landowner with foreknowledge of their type. With no adverse selection or moral hazard, both IR constraints are binding (Laffont and Tirole 1988), and the optimal first-best (denoted by superscript FB) payments are given by:

$$F_i^{FB} = c_i(x_i^{FB}) \forall i \in \{l, h\} \tag{4.5}$$

Substituting (5) into (1) gives the following optimisation problem for the first-best input allocation:

$$\max_{x_l, x_h} \mathbb{E}_\phi[\sum_{i \in \{l,h\}} v(x_i) - (1+\lambda)c_i(x_i)] \tag{4.6}$$

Given that only $v(x)$ is stochastic, Equation (4.6) is equivalent to:

$$\max_{x_l, x_h} \sum_{i \in \{l,h\}} \sum_{j=1}^{n} a_j(\phi_j - \phi_0)x_i + \phi_0 x_i - (1+\lambda)c_i(x_i) \tag{4.7}$$

Note that $\mathbb{E}_\phi[(v(x_i))] = \sum_{j=1}^{n} a_j(\phi_j - \phi_0)x_i + \phi_0 x_i$. Equation (4.7) can be used to derive conditions for the first-best input allocations, x_l^{FB} and x_h^{FB}:

$$c_l^{'}(x_l^{FB}) = \frac{\sum_{j=1}^{n} a_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} \tag{4.8}$$

$$c_h^{'}(x_h^{FB}) = \frac{\sum_{j=1}^{n} a_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} \tag{4.9}$$

The term $\sum_{j=1}^{n} a_j(\phi_j - \phi_0) + \phi_0$ is the expected value of the per-unit input ES produced. Equations (4.8) and (4.9) show that an increase in any a_j, indicating a higher probability of achieving a particular level of ES above the baseline, raises the optimal marginal conservation cost (as by assumption $\phi_j - \phi_0 > 0 \forall j$), and thus increases the optimal input (as $c_i^{'}(x)$ is increasing in $x \forall i$); the same is true for a higher $\phi_j \forall j$. An increase in λ corresponds to greater economic inefficiency (deadweight loss) arising from the allocation of resources to conservation efforts, and therefore a decrease in the optimal input. Finally, as type-h landowners have higher conservation costs, Equations (4.8) and (4.9) show that when the marginal benefits of conservation are the same for both types, the optimal input will be greater for the type-l.

However, the first-best is not implementable when landowners have private information of their type. In this instance, a type-l landowner is incentivised to mimic a type-h landowner (i.e., choose that contract), and collect an information rent:

$$F_h - c_l(x_h) = c_h(x_h) - c_l(x_h) \geq 0 \tag{4.10}$$

In the following section, we examine the second-best contract with adverse selection using the principal-agent framework.

4.3.2 Adverse Selection

We assume that the principal does not know a specific landowner's type, but does know the distribution over types (Ozanne and White 2007; Mason and Plantinga 2013). The principal's objective function is now:

$$\max_{x_l, x_h, F_l, F_h} \mathbb{E}_\phi[\sum_{i \in \{l,h\}} \gamma_i \Big(v(x_i) + (F_i - c_i(x_i)) - (1+\lambda)F_i \Big)] \tag{4.11}$$

Where γ_i refers to the likelihood of a landowner being type-i. Further, in addition to the IR_i constraints, the principal must contend with the following IC_i constraints:

$$F_i - c_i(x_i) \geq F_j - c_i(x_j) \forall i, j, i \neq j \tag{4.12}$$

The IC_i constraints ensure that the contract for a type-i landowner is designed so that each prefers the contract intended for their own type over that of the other. As shown by Equation (4.10), only the type-l landowner is incentivised to mimic the type-h. Hence, the IR_h and IC_l constraints are binding (Ozanne and White 2007; White and Hanley 2016), and the optimal second-best payments (denoted by superscript SB) are characterised by:

$$F_h^{SB} = c_h(x_h^{SB}) \tag{4.13}$$

$$F_l^{SB} = c_h(x_h^{SB}) - c_l(x_h^{SB}) + c_l(x_l^{SB}) \tag{4.14}$$

Substituting Equations (4.13) and (4.14) into Equation (4.11) gives the following optimisation problem:

$$\max_{x_l,x_h} \sum_{i\in\{l,h\}} \gamma_i \Big(\sum_{j=1}^{n} a_j(\phi_j - \phi_0) + \phi_0 - (1+\lambda)c_i(x_i) \Big) - \gamma_l \lambda (c_h(x_h) - c_l(x_h)) \tag{4.15}$$

Taking first-order conditions yields the equations that describe the second-best input allocation with adverse selection:

$$c_l^{'}(x_l^{SB}) = \frac{\sum_{j=1}^{n} a_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} \tag{4.16}$$

$$c_h^{'}(x_h^{SB}) = \frac{\sum_{j=1}^{n} a_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \frac{\gamma_l \lambda}{\gamma_h (1+\lambda)} [c_h^{'}(x_h^{SB}) - c_l^{'}(x_h^{SB})] \tag{4.17}$$

It must be the case that $\frac{\gamma_l \lambda}{\gamma_h (1+\lambda)} [c_h^{'}(x_h^{SB}) - c_l^{'}(x_h^{SB})] \geq 0$ (Ozanne and White 2007). Adverse selection does not result in any output distortion with respect to the less-efficient landowner, $x_l^{SB} = x_l^{FB}$. However, there is a downward distortion to input with respect to the type-h, motivated by a desire to reduce the information rent collected by the type-l at the expense of production efficiency (e.g., Laffont and Tirole 1988). Equations (4.16) and (4.17) replicate well-established findings in the contract literature. In the next section, we extend the analysis to explore how uncertainty impacts contract design in PES.

4.4 Robust Contract

4.4.1 First-Best Allocation

Uncertainty is incorporated into the model by altering the formulation of the ES production function in Equation (4.2). Assume that the principal does not know $(a_1, ..., a_n)$ perfectly, but instead knows that the probabilities of outcomes lie within the following uncertainty set. This can be

expressed as:

$$\mathcal{U} = \{(\bar{a}_1, ..., \bar{a}_n) + (\delta_1 u_1, ..., \delta_n u_n) | ||\mathbf{u}|| \leq 1\} \tag{4.18}$$

Where δ_j must be such that $\bar{a}_j - \delta_j > 0 \forall j$ and $\sum_j \delta_j^2 < (\sum_j \bar{a}_j - 1)^2$ to ensure non-negative probabilities (Yu and Kong 2020). The principal and landowner are both aware of the range of possible outcomes but lack certainty about the probabilities associated with each, and neither has knowledge of the specific characteristics of $\mathbf{u}$. This reflects the current state of knowledge about ES. For instance, Costanza et al. (1997) famously estimated the global value of ecosystem services to range between \$16 and \$54 trillion (in 1997 USD), with this broad variation in estimates stemming from differing model assumptions.

Lacking knowledge of the specific characteristics of uncertainty, the principal wishes to design a contract that ensures a positive return is guaranteed under all possible realisations of $\mathbf{u}$. A simple way to obtain this performance guarantee is to design a scheme that optimises for the worst-case outcome in the uncertainty set. The uncertainty-averse principal does this by having max-min preferences, and replacing a problem with infinitely many probabilistic constraints with a single deterministic one (Bertsimas et al. 2011). This problem can be expressed as:

$$\max_{x_l, x_h, F_l, F_h} \{\min_{\mathbf{a}} \mathbb{E}_\phi [\sum_{i \in \{l,h\}} v(x_i) - c_i(x_i) - \lambda F_i]\} \tag{4.19}$$

Landowners are unaffected by uncertainty when payments are input-based. Thus, there is no change to the IR_i constraints introduced in Equation (4.3); each constraint remains binding in the robust first-best (denoted by superscript R, FB). Hence, $F_i^{R,FB} = c_i(x_i^{R,FB}) \forall i$. The social welfare maximisation problem can be reformulated as:

$$\max_{x_l, x_h} \sum_{i \in \{l,h\}} \Big(\min_{\mathbf{a}} \{\sum_{j=1}^{n} a_j(\phi_j - \phi_0) x_i\} + \phi_0 x_i - (1+\lambda) c_i(x_i) \Big) \tag{4.20}$$

Ben-Tal and Nemirovski (1999) show that, given two vectors $\mathbf{a}, \mathbf{x}$, where $\mathbf{a}$ represents coefficient vectors in the ellipsoidal uncertainty set $\mathcal{U}$, we have that $U(\mathbf{x}) = \min_{a \in \mathcal{U}} \langle \mathbf{a}, \mathbf{x} \rangle =$

$\sum_j \bar{a}_j x_j - \sqrt{\sum_j \delta_j^2 x_j^2}$, and that $\sqrt{\sum_j \delta_j^2 x_j^2}$ is convex (see Section 4.7.1 in the Appendix for more details). Yu and Kong (2020) show that $U(\mathbf{x})$ is concave, and is increasing in $\mathbf{x}, \forall \mathbf{x} > 0$. These properties imply that the optimisation problem has a solution. The term $\min_{\mathbf{a}} \sum_{j=1}^{n} a_j(\phi_j - \phi_0)x_i$ can thus be expanded to produce $\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0)x_i - \sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2 x_i^2}$. Hence, Equation (4.20) can be expressed as a standard optimisation problem:

$$\max_{x_l, x_h} \sum_{i \in \{l,h\}} \sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0)x_i - \sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2 x_i} + \phi_0 x_i - (1+\lambda)c_i(x_i) \tag{4.21}$$

Equation (4.21) describes the social welfare function for an uncertainty-averse principal with max-min preferences. Optimising with respect to this expression brings us the optimal robust input allocations, $x_l^{R,FB}, x_h^{R,FB}$:

$$c_l^{'}(x_l^{R,FB}) = \frac{\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \frac{\sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}}{(1+\lambda)} \tag{4.22}$$

$$c_h^{'}(x_h^{R,FB}) = \frac{\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \frac{\sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}}{(1+\lambda)} \tag{4.23}$$

It must be the case that $\sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2} > 0 \forall j$, as $\phi_j - \phi_0 \geq 0 \forall j$. Uncertainty reduces the terms on the right-hand side of Equations (4.22) and (4.23) relative to the first-best. As $c_i^{'}(x_i)$ is increasing in x_i, an increase in $\delta_j \forall j$ must be associated with a decrease in the contracted input. This conservative adjustment increases with greater uncertainty (see that $c_i^{'}(x_i^{R,FB})$ is decreasing in δ_j). Finally, with decreasing uncertainty, where $\delta_j \to 0 \forall j$, which can be interpreted as increasing confidence on the part of the principal, the contracted land input approaches the first-best in Section 4.3. In the following section, we describe how the optimal contract deviates from the robust first-

best given that the landowner has private information on their type.

4.4.2 Adverse Selection

This section re-introduces adverse selection to characterise the robust second-best contract. The principal's probability assessment for each landowner type must be incorporated into Equation (4.19), resulting in:

$$\max_{x_l,x_h,F_l,F_h} \min_{\mathbf{a}} \{ \sum_{i\in\{l,h\}} \gamma_i \Big(\sum_{j=1}^{n} a_j(\phi_j - \phi_0)x_i - c_i(x_i) - \lambda F_i \Big) \} \tag{4.24}$$

Again, uncertainty with respect to the benefits generated by conservation has no impact on the type-constraints introduced in Section 4.3.2. IC_l and IC_h remain binding, and $F_l^{R,SB} = c_h(x_h^{R,SB}) - c_l(x_h^{R,SB}) + c_l(x_l^{R,SB}), F_h^{R,SB} = c_h(x_h^{R,SB})$, where the superscript R, SB defines the robust second-best. Equation (4.24) can thus be re-expressed as:

$$\max_{x_l,x_h} \sum_{i\in\{l,h\}} \gamma_i \Big(\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0)x_i - \sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2 x_i} + \phi_0 x_i - (1+\lambda)c_i(x_i) \Big) - \gamma_l \lambda (c_h(x_h) - c_l(x_h)) \tag{4.25}$$

Standard optimisation leads to the following proposition:

Proposition 1 *Uncertainty regarding the benefits of conservation does not alter the optimal payment structure of an input-based PES contract. Both with and without adverse selection, landowners continue to participate as long as they are compensated for their conservation costs. As only the principal is exposed to uncertainty, there is a downward distortion to the contracted input. In both cases, the principal adopts a conservative approach, reducing investment in ES to secure a performance guarantee to ensure positive returns in the worst-case scenario of ES provision. With adverse selection, the contract still only compensates the type-h landowner up to their costs, while the type-l collects an information rent. This results in an additional downward distortion in put for the more efficient landowner.*

The optimal robust second-best input allocations, $x_l^{R,SB}, x_h^{R,SB}$, are given by taking first-order conditions with respect to Equation (4.25):

$$c_l^{'}(x_l^{R,SB}) = \frac{\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \frac{\sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}}{(1+\lambda)} \tag{4.26}$$

$$c_h^{'}(x_h^{R,SB}) = \frac{\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \frac{\sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}}{(1+\lambda)} - \frac{\gamma_l \lambda}{\gamma_h(1+\lambda)}(c_h^{'}(x_h^{R,SB}) - c_l^{'}(x_h^{R,SB})) \tag{4.27}$$

Where $c_l^{'}(x_l^{R,SB}) < c_l^{'}(x_l^{SB})$, $c_h^{'}(x_h^{R,SB}) < c_h^{'}(x_h^{SB})$. Proposition 1 describes the actions an uncertainty-averse principal can take to shield themselves from worst-case scenarios. The principal is more conservative about investing in ES in both the robust first- and second-best. For the type-h landowner, input is further reduced due to the adverse selection adjustment (represented by $\frac{\gamma_l \lambda}{\gamma_h(1+\lambda)}(c_h^{'}(x_h^{R,SB}) - c_l^{'}(x_h^{R,SB}))$).

4.4.3 Numerical Simulation

We have shown that the presence of uncertainty changes the nature of optimal contracts in PES. The extent to which first-best and second-best allocations diverge from their robust counterparts will depend on the degree of uncertainty in practice. In this section, we illustrate our findings via a numerical exercise and simulation using parameters derived for landowners and watershed services in Canada.

For the numerical exercise, we adapt the set-up of Ozanne et al. (2001) to ensure landowner costs and ES are represented in a credible manner. The ES production function takes the form:

$$v(x) = \phi x, \phi \sim \text{Uniform}(a, b) \tag{4.28}$$

The findings of the watershed valuation exercise on Canada's drinking water treatment facilities in Chapter 2 are used to represent ϕ. The per-hectare monetary value associated with increases in forest cover estimated using the 2015 Survey of Drinking Water Plants (SDWP) in Canada is \$18.83 CAD/ha/yr. To represent ES probabilistically, a 90% confidence interval is constructed around this mean estimate[4], and 20 points are drawn from a uniform distribution within these bounds. These points are assigned an equal probability of occurrence.

The profit function for a type-i landowner is assumed to take the following form:

$$\pi_i(x_i) = P_i A_i x_i^{\beta} - R_i x_i, i \in \{l, h\} \tag{4.29}$$

Where P_i, R_i, and A_i are output prices, input prices, and technical efficiency for a type-i landowner. β represents the partial elasticity of output for agriculture in Canada. In order to capture heterogeneity, two landowner types are included: soybean and grain corn cultivators. Ontario is a major producer of the two crops, producing roughly 60% of the nation's output (Ontario Ministry of Agriculture, Food and Rural Affairs 2024).

To obtain parameters for conservation costs (described in Table 4.1), we leverage data from Statistics Canada and the 2023 Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA) Field Crop Budget, an annual resource produced for landowners in the province. The price of each crop per ton produced in 2023 is taken as the average of monthly prices in Ontario from Statistics Canada Table 32-10-0077-01 (*Farm produce prices, crops and livestock*). 2023 seed costs are used to represent total input costs for each landowner. The Field Crop Budget further contains

[4]This set-up is simple for the purposes of an illustrative example, see Bertsimas et al. (2018) for a more data-driven approach to constructing uncertainty sets in practice.

information on the average yield per hectare in tonnes, included here as a measure of technical efficiency.

Estimates for γ_i, the probability of a landowner being of type-i, are derived from crop density maps produced by Agriculture and Agri-Food Canada. Specifically, each pixel in the raster map for a crop indicates its level of spatial density in that location. Regions with a higher spatial density have a higher likelihood of that crop having been planted there. The probability of a landowner cultivating soybeans is taken as the proportion of land where the spatial density of soybeans surpasses that of corn, the remaining probability is assigned to grain corn cultivation.

Table 4.1: Summary of Simulation Parameters

Variable Type	**Source**	**Symbol**	**Value**
Price of soybeans in Ontario, 2023 ($CAD/t)	Statistics Canada Table 32-10-0077-01	P_h	702.09
Price of corn in Ontario, 2023 ($CAD/t)	Statistics Canada Table 32-10-0077-01	P_l	298.68
Cost of producing soybeans in Ontario, 2023 ($CAD/ha)	OMAFRA Field Budgets, 2023	R_h	150.61
Cost of producing corn in Ontario, 2023 ($CAD/ha)	OMAFRA Field Budgets, 2023	R_l	319.01
Tonnes of soybeans produced in Ontario, 2023 ($t/ha)	OMAFRA Field Budgets, 2023	A_h	3.29
Tonnes of corn produced in Ontario, 2023 ($/ha)	OMAFRA Field Budgets, 2023	A_l	11.32
Partial elasticity of output to land	Tabeau et al. (2017)	β	0.60

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Table 4.1 continued from previous page

Variable Type	Source	Symbol	Value
Probability type-l landowner	Spatial Density of Major Crops - Agriculture and Agri-Food Canada	γ_l	0.55
Probability type-h landowner	Spatial Density of Major Crops - Agriculture and Agri-Food Canada	γ_h	0.45
Deadweight loss factor	Ozanne et al. (2001)	λ	0.1

In representing uncertainty, for simplicity we assume that $\delta_j = \delta \forall j$, and generate a sequence of values δ can take that meet the conditions described in the preceding section to systematically explore the impact varying degrees of uncertainty have on robust PES contracts and performance. Given this set-up, Table 4.2 presents findings for the optimal input allocations and payments with no uncertainty.

Table 4.2: Numerical estimates of optimal policy

Scenario	Land in agriculture (ha)	Land conserved (ha)	Compensation payment ($ CAD/ha)
Type-*l*			
No policy	234.34	0	0
First-best	175.94	58.39	8.65
Second-best	175.94	58.39	9.95
Type-*h*			
No policy	94.29	0	0
First-best	82.02	12.27	8.97
Second-best	82.95	11.34	8.24

The results in Table 4.2 align well with theoretical expectations. With no policy, type-l landowners allocate 234.34 hectares to soybean cultivation, while type-h landowners allocate 94.29 hectares to corn. At the first-best, both conserve. Type-l landowners protect 58.39 hectares of forest and re-

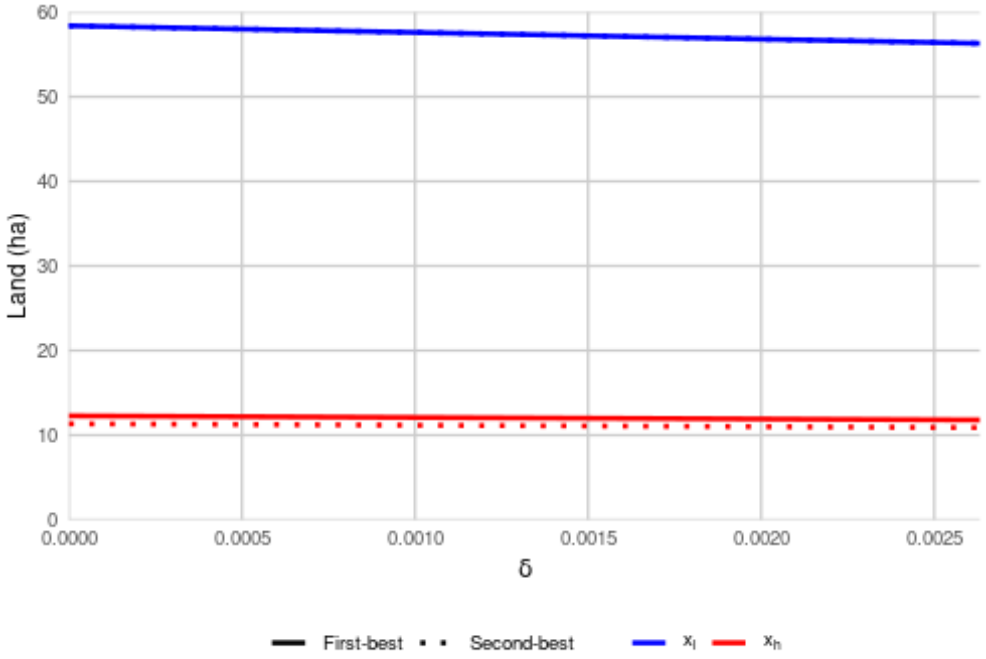


Figure 4.1: First-Best and Second-Best Robust Input Allocation

ceive a payment of $8.65 CAD/ha, while type-$h$ landowners conserve less, at only 12.27 hectares, earning $8.97 CAD/ha. In the second-best, where landowners have private information regarding their types, contracted input remains the same as in the first-best for type-l landowners. However, their compensation increases to $9.95 CAD/ha to induce truthful self-reporting of type. Conversely, type-h landowners are contracted for less input, with a corresponding reduction in the optimal payment. This reduction reflects the impact of adverse selection. There is an efficiency loss that is counter-weighed by a gain from minimising the information rent extracted by the less efficient type.

Figure 4.1 presents the impact of uncertainty on the first- and second-best robust input allocations for each landowner type. Increasing uncertainty is denoted by a larger δ. As the degree of uncertainty increases, the optimal robust first-and second-best allocation declines for both types. This reflects the principal's willingness-to-invest in ES provision declining as the benefits of conservation become less certain. As expected, there is again no difference between the robust first- and second-best input allocation for the type-l, while for the type-h, the presence of adverse selec-

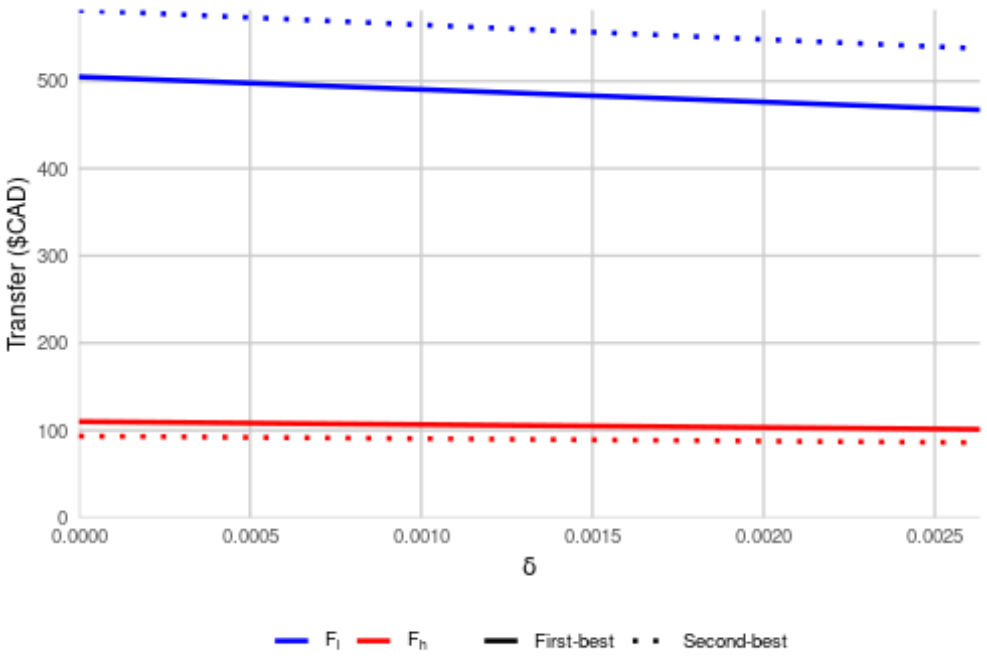


Figure 4.2: First-Best and Second-Best Robust Payments

tion leads to a downward output distortion.

Figure 4.2 presents optimal payments with increasing uncertainty for each landowner type in the first- and second-best. For both landowner types, payments decrease in uncertainty, which is to be expected given the impact of uncertainty on contracted input. The output distortion induced by adverse selection results in a lower compensation for the type-h landowner in the second-best. Conversely, type-l landowners again enjoy higher payments in the robust second-best as the principal seeks to incentivise truthful self-reporting.

Finally, Figure 4.3 displays the results of a simple Monte Carlo simulation procedure. For each δ, over 5000 runs, a vector $\mathbf{u} \in \{-1, 1\}$ is randomly generated, with each run representing a realisation of the "true" distribution of ES within a given uncertainty set. The figure shows the average expected social welfare over the realised distributions given optimal inputs and payments for either method with and without adverse selection. Expected social welfare is represented by Equations (4.1) and (4.11) for the cases with and without adverse selection, respectively. When δ is low, the distortion to the expected input is low as well. For this reason, the two contracts initially

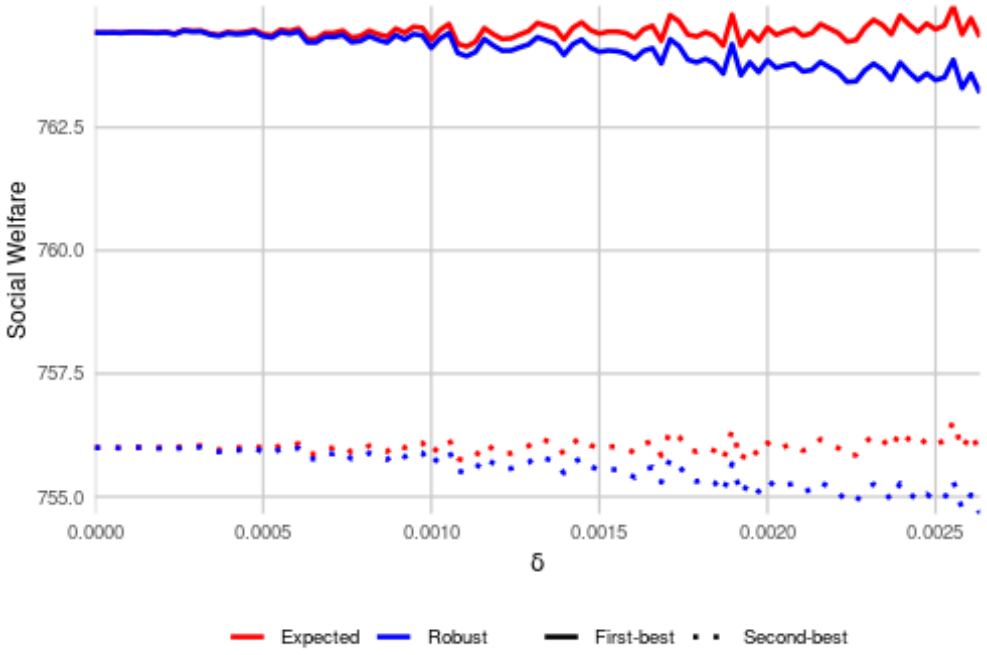


Figure 4.3: Expected Social Welfare Once Distribution Is Realised

result in near-identical expected outcomes. However, as uncertainty increases, two trends become evident. First, expected social welfare becomes increasingly variable in uncertainty. Secondly, with larger values of δ, a divergence emerges in the average expected social welfare for each contract. The simulation essentially describes the social welfare costs of uncertainty-aversion. By conservatively investing in ES, the principal reduces exposure to worst-case scenarios by limiting risks associated with extreme realisations of uncertainty. At the same time, this conservatism implies a trade-off: it prioritises robustness but does so at the cost of foregoing opportunities for higher gains that could arise under more favorable conditions (Gorissen et al. 2015). Essentially, if the true probability distribution assigns higher likelihoods to marginal land productivity above the baseline compared to the principal's worst-case assumption, a robust contract will under-invest in conservation relative to the standard contract.

4.5 Moral hazard

4.5.1 Input-based payments

We now introduce moral hazard to the conventional problem with a representative risk-neutral landowner and a social-welfare maximising principal. Moral hazard is often of greater relevance to principals than adverse selection in practice (Hart and Latacz-Lohmann 2005). The following *effort-contingent* probability distribution is assigned to ES production:

$$v(x,e) = \begin{cases} \phi_n x & \text{w.p.} a_n e \\ \phi_{n-1} x & \text{w.p.} a_{n-1} e \\ \vdots & \\ \phi_1 x & \text{w.p.} a_1 e \\ \phi_0 x & \text{w.p.} 1 - \sum_{j=1}^{n} a_j e \end{cases} \tag{4.30}$$

Effort is a new feature in the model, a second input alongside land, with $e \in \{e_1, e_2\}, e_2 > e_1$. Essentially, effort increases the likelihood of higher output above the baseline (ϕ_0). Landowners can exert effort by, for example, introducing best management practices (BMPs) that enhance ES provision. The PES literature shows that landowners are often contracted for the implementation of BMPs (e.g., short cycle or permanent cover crops; see Kosoy et al. 2007) in addition to to conservation. Landowner conservation costs are also now a function of both land and effort. For simplicity, the cost function takes the form:

$$C(x,e) = \pi(x^*) - \pi(x^* - x) + \psi(x,e) = c(x) + \psi(x,e) \tag{4.31}$$

Where $\psi(x,e)$ represents the cost to the landowner of exerting effort. $\psi(x,e)$ is such that $\frac{\partial \psi}{\partial x}, \frac{\partial \psi}{\partial e}, \frac{\partial^2 \psi}{\partial x_i^2}, \frac{\partial^2 \psi}{\partial e^2} > 0$. In many circumstances, effort is unobservable. When this is the case,

principals can induce high effort by introducing a variable payment structure, with agents earning more as output increases. This treatment is particularly common in the agency literature (e.g., Carroll 2015), and we will explore the implications of such a structure on PES in the next section. Here, effort is assumed to be verifiable but costly (e.g., Ozanne and White 2007), with the cost of monitoring, $M(p)$ assumed increasing in p, the monitoring rate. The principal's optimisation problem with moral hazard and effort can be represented as:

$$\max_{x,F,p} \mathbb{E}_{\phi}[v(x, e_2) + (F - C(x, e_2)) - (1 + \lambda)(F + M(p))] \tag{4.32}$$

We assume that it is always worthwhile to contract landowners on effort. This is true when e_2 is sufficiently large to offset the additional costs of monitoring and inducing effort. In addition to the deadweight loss from raising public funds for payments, the principal must contend with the social cost of monitoring, represented by $(1 + \lambda)M(p)$. Equation (4.32) is subject to the following IR constraint to ensure a non-negative payoff for the landowner:

$$F - C(x, e_2) \geq 0 \tag{4.33}$$

Clearly, the IR constraint is binding, and $F^{I,MH} = C(x^{I,MH}, e_2)$, where the superscript I, MH indicates the optimal input-based solution with moral hazard. However, given that effort is costly to verify and the landowner has an incentive to shirk, there is also a moral hazard incentive compatibility ($MHIC$) constraint:

$$F - C(x, e_2) \geq p(F - C(x, e_1) - \eta) + (1 - p)(F - C(x, e_1)) \tag{4.34}$$

The left-hand side of the $MHIC$ constraint describes the net payoff to the landowner from exerting high effort. On the right-hand side is the net expected payoff from being contracted to

exert effort and shirking. With probability p, non-compliance is detected, and the landowner surrenders the penalty η. With probability $(1-p)$, their compliance goes undetected, and they earn the contracted payment without exerting effort. We assume a fixed fine for the landowner when shirking is detected. If the principal has unlimited flexibility in setting p and η, the optimal monitoring program for a risk-neutral landowner consists of a harsh punishment for shirking with a very low monitoring rate to ensure expected penalties are high enough to deter non-compliance without ever having to monitor, a "*Mirrlees shoot the Agent*" contract (Georgiadis and Szentes 2020). Previous literature has discussed the inflexibility of fines as a policy tool for principals. Fines are typically low relative to payments in practice as a result of limited liability for landowners; we assume fines are fixed by the judicial system and the rate of monitoring is determined by the principal (Swierzbinski 1994; Choe and Fraser 1999; Ozanne et al. 2001; Hart and Latacz-Lohmann 2005).

Lemma 4.5.1 *Where landowners are risk-neutral, the $MHIC$ constraints is binding. Further, they can be simplified to produce:*

$$p^{I,MH} = \frac{\Delta\psi(x^{I,MH})}{\eta} \tag{4.35}$$

Where $\Delta\psi(x) = \psi(x^{I,MH}, e_2) - \psi(x^{I,MH}, e_1)$.

The proof for Lemma $4.5.1$ can be found in Section 4.7.4 of the Appendix. For simplicity, we normalise $\psi(x, e_1) = 0$, which corresponds to the landowner not enacting any additional BMPs as contracted, and thus bearing no additional cost, and express $\psi(x, e_2) = \psi(x)$. Using Equations (4.33) and (4.35), Equation (4.32) can be simplified to produce:

$$\max_x \mathbb{E}_\phi[v(x) - (1+\lambda)\Big(C(x, e_2) + M(\frac{\psi(x)}{\eta})\Big)] \tag{4.36}$$

From Equation (4.36), the second-best land allocation with moral hazard when payments are input-based and effort is verifiable but costly, $x^{I,MH}$, is characterised by the following relationship:

$$c^{'}(x^{I,MH}) = \frac{\sum_{j=1}^{n} a_j e_2(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \psi^{'}(x^{I,MH})\Big(1 + \frac{1}{\eta} M^{'}(\frac{\psi(x^{I,MH})}{\eta})\Big) \tag{4.37}$$

Moral hazard also induces a downward distortion to land input, reflecting the costliness of compliance monitoring as captured by $M(p)$. As the cost of monitoring increases, overall expenses rise, leading to a reduction in input. It becomes clear from Lemma 4.5.1 as well that, were the principal able to choose both p and η, the optimal monitoring scheme for a risk-neutral landowner involves a penalty that is set very high and a monitoring rate that is close to zero - as $\eta \to \infty, \frac{1}{\eta} \to 0$, and Equation (4.37) is equivalent to the conditions in Equations (4.8) and (4.9) with a single landowner.

4.5.2 Input-based payments with uncertainty

Moral hazard alters one of the constraints on the uncertainty set in Equation (4.18), δ_j must be such that $\sum_j \delta_j^2 < (\sum_j \bar{a}_j^2 - \frac{1}{e_2})^2$ to ensure that probabilities of events are always non-negative (Yu and Kong 2020). The principal's objective with a single landowner in the presence of uncertainty is:

$$\max_{x,F,p}\{\min_{\mathbf{a}} \mathbb{E}_{\phi}[v(x,e_2) - (F - C(x,e_2)) + (1+\lambda)(F + M(p))]\} \tag{4.38}$$

Note that uncertainty does not impact the IR or $MHIC$ constraints, and both remain binding, so that $F^{I,R,MH} = C(x^{I,R,MH}, e_2)$ and $p^{I,R,MH} = \frac{\Delta\psi(x^{I,R,MH})}{\eta}$, where the superscript I, R, MH indicates the optimal robust input-based solution with moral hazard. Equation (4.38) can be rewritten as:

$$\max_{x} \min_{\mathbf{a}} \{\sum_{j=1}^{n} a_j e_2(\phi_j - \phi_0)x + \phi_0 x - (1+\lambda)\Big(c(x,e_2) + M(\frac{\psi(x)}{\eta})\Big)\} \tag{4.39}$$

Taking first-order conditions leads to the following Proposition:

Proposition 2 *The optimal contract with moral hazard and uncertainty compensates the landowner just up to their conservation costs, and again induces compliance with monitoring and penalties, resulting in a distortion to land input. However, there is an additional downward distortion to land input induced by uncertainty, as the principal reduces investment into ES provision to improve their payoff in the worst-case scenario.*

The optimal robust land allocation with moral hazard and input-based contracts, $x^{I,R,MH}$, when effort is verifiable but costly, is given by:

$$c^{'}(x^{I,R,MH}) = \frac{\sum_{j=1}^{n} \bar{a}_j e_2(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \frac{e_2\sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}}{(1+\lambda)} - \psi^{'}(x^{I,R,MH})\Big(1 + \frac{1}{\eta}M^{'}(\frac{\psi(x^{I,R,MH})}{\eta})\Big) \tag{4.40}$$

Where $c^{'}(x^{I,R,MH}) < c^{'}(x^{I,MH})$. Proposition 2 expresses that there is again a downward distortion in the optimal input level by the uncertainty-averse adjustment factor. The risk-adjustment term does not interact with the negative moral hazard distortion from Section 4.5.1, as it did not with the adverse selection-driven distortion in Section 4.4.2. Given input-based payments and the assumption that conservation costs are known with certainty, the landowner problem is not impacted by uncertainty. That conservation costs are known with certainty seems reasonable (e.g., a landowner should be able to learn the expenses associated with specific conservation practices before participating in a scheme). However, when the conservation costs include agricultural profits foregone, this assumption may be too restrictive, as crop yield is also impacted by unpredictable variables like precipitation and temperature. Landowners prefer PES contracts with more predictable, stable payments over uncertain, variable payments (Bartkowski et al. 2021). In the next section, we describe the moral hazard problem in a setting where the optimal payment structure includes variable, output-based payments. Further, we show that the optimal contract will include

a bonus payment structure increasing with ES output to account for risky and uncertain returns to the landowner.

4.5.3 Output-based payments

In this section, the moral hazard problem is characterised differently. Assume that effort is not verifiable, or that verification is too costly for the principal, but ES output is known with certainty once realised. In the context of watershed ES, monitoring is particularly challenging for inputs that are unobservable, e.g., the management of fertilisers, pesticides, and herbicides (OECD 2022). In such circumstances, if outcomes are measurable (e.g., water quality at source water intake), the principal can employ variable, output-based payments to induce effort. While few PES schemes employ output-based payments today, they are of increasing interest, driven by a growing sentiment that schemes with input-based payments are failing to deliver results. Little discussion has been given to the complications that may arise due to uncertainty. In this section, the model is extended to incorporate output-based payments to describe the influence this has on contracts. With output-based payments, the principal's objective function given a representative landowner takes the form:

$$\max_{x,\mathbf{F}} \mathbb{E}_{\phi}[v(x) - C(x, e_2) - \lambda F)] \tag{4.41}$$

Where $\mathbf{F} = \{F_j, j = 0, 1, ..., n\}$ represents a vector of payments indexed by each possible realization of the ES output ϕ. Each element F_j corresponds to a payment that is contingent on a particular realized state j, allowing the principal to condition payments on the level of ES provided. Unlike in previous sections, where the compensation F remains constant regardless of actual ES output, output-based payments vary with realised outcomes. This structure incentivises the landowner by aligning compensation more closely with the actual provision of ES. Equation (4.41) is equivalent to:

$$\max_{x,\mathbf{F}} \sum_{j=1}^{n} a_j e_2 \Big((\phi_j - \phi_0)x - \lambda(F_j - F_0) \Big) + \phi_0 x - \lambda F_0 - C(x, e_2) \tag{4.42}$$

To move from Equation (4.41) to (4.42), the expected output $\mathbb{E}_\phi[v(x)]$ is expressed in terms of the weighted sum $\sum_{j=1}^{n} a_j e_2(\phi_j x)$, where a_j represents the probability of each state j occurring, and e_2 reflects the effort level chosen by the landowner. The terms are then reorganised by factoring out the baseline output ϕ_0 and the baseline payment F_0 to isolate deviations from these baselines in terms of each state-specific component $(\phi_j - \phi_0)$ and $(F_j - F_0)$, simplifying the expression for optimisation. Social welfare is maximised subject to the following $n + 1$ IR_j constraints:

$$F_j \geq C(x, e_2) \forall j = 0, 1, ..., n \tag{4.43}$$

Each IR_j constraint ensure that the landowner's payoff is non-negative given that state j is realised. As effort is unverifiable, the principal can no longer use monitoring and the threat of sanctions to induce compliance from the landowner. They instead incentivise exerting effort by offering bonus payments to the landowner when output is higher than the baseline. The $MHIC$ is now:

$$\sum_{j=1}^{n} a_j e_2 F_j + (1 - \sum_{j=1}^{n} a_j e_2) F_0 - C(x, e_2) \geq \sum_{j=1}^{n} a_j e_1 F_j + (1 - \sum_{j=1}^{n} a_j e_1) F_0 - C(x, e_1) \tag{4.44}$$

The $MHIC$ in Equation (4.44) ensures that the expected payoff from exerting e_2 is at least as large as the expected payoff from e_1. Here, a_j represents the probability associated with each output state j, while the effort level e affects these probabilities by altering the likelihood of each output level being realized. When the landowner exerts high effort e_2, it increases the probability of higher output levels, impacting the expected value of payments F_j in each state. Conversely,

with lower effort e_1, the expected payoff decreases, as reflected in the second term. From Equation (4.43), it is immediately clear that IR_0 must be binding:

$$F_0^{O,MH} = C(x^{O,MH}, e_2) \tag{4.45}$$

Where the superscript O, MH indicates the optimal solution for output-based payments with moral hazard. In other words, the minimum payment the principal provides to the landowner must ensure that participation in the contract is incentivised even given the worst-case realisation of ϕ_0. Assuming the cost function is still described by Equation (4.31), the $MHIC$ constraint can be rearranged to produce:

$$\sum_{j=1}^{n} a_j w_j(x) \geq \frac{\psi(x)}{(e_2 - e_1)} \tag{4.46}$$

Where $w_j(x) = F_j - F_0 = F_j - C(x, e_2)$ (this derivation can be found in Section 4.7.5 of the Appendix). As was the case in Sections 1 and 2, increases in payments to the landowner are unambiguously associated with reductions in social welfare, and Equation (4.46) must be binding. The $MHIC$ constraint means that a uniform payment to the landowner can never be optimal in the setting with output-based payments, as that would imply that $w_j(x) = 0 \forall j$. Substituting the terms in Equations (4.45) and (4.46) into Equation (4.42) gives:

$$\max_{x, \mathbf{w}(\mathbf{x}) \geq 0} \sum_{j=1}^{n} a_j e_2(\phi_j - \phi_0)x - \lambda e_2 \frac{\psi(x, e_2)}{(e_2 - e_1)} + \phi_0 x - (1 + \lambda)C(x, e_2) \tag{4.47}$$

In Equation (4.47), the term $\frac{\psi(x,e_2)}{e_2 - e_1}$ replaces $\sum_j a_j(F_j - F_0)$ from Equation (4.42). This substitution is based on the moral hazard incentive compatibility constraint ($MHIC$) in Equation (4.46), which ensures that the additional cost $\psi(x, e_2)$ of exerting high effort is covered by expected bonus

payments. This replacement expresses the necessary incentive cost directly in terms of $\psi(x, e_2)$, simplifying the objective function while maintaining the compliance condition. Notice that $\mathbf{w}(\mathbf{x})$ does not feature in the objective function in Equation (4.47). This leads us to Proposition 3:

Proposition 3 *When effort is not verifiable but output is measurable, a hybrid payment structure is optimal. Landowners are compensated up to their conservation costs to induce participation. There are additional bonus payments that increase in output to ensure landowners exert effort. Any variable payment structure is optimal as long as the expected value of bonuses exceeds the expected cost of exerting effort. The principal thus motivates the landowner to exert effort by providing higher payments when higher output is realized. There is no unique solution for contracts. However, the necessity of providing bonus payments to induce compliance results in a downward distortion to land input.*

With output-based payments, any variable payment structure, $\mathbf{w}(\mathbf{x})$ is optimal, provided that the base payment covers the landowner's costs of compliance and that the $MHIC$ is met by the bonus payments. Further, the solution for the optimal land input with output-based contracts is:

$$c^{'}(x^{O,MH}) = \frac{\sum_{j=1}^{n} a_j e_2(\phi_j - \phi_0) + \phi_0}{(1+\lambda)} - \psi^{'}(x^{O,MH})\Big(1 + \frac{\lambda e_2}{(e_2 - e_1)(1+\lambda)}\Big) \tag{4.48}$$

Proposition 3 shows that the use of output-based payments (as opposed to monitoring and sanctioning) still results in a downward distortion from the first-best. Whether this distortion is greater than in the case with input-based payments depends on the value of the last term in Equation (4.48), which represents the social cost of raising funds to induce effort from the landowner. When $\frac{\lambda e_2}{(e_2-e_1)(1+\lambda)} > \frac{1}{\eta} M'(\frac{\psi(x)}{\eta})$, the distortion will be larger for the output-based payment structure. Note that the term is decreasing in e_1, indicating that as the potential for effort to increase production decreases, the optimal input allocation also decreases.

4.5.4 Output-based payments with uncertainty

The preceding section found that there is no unique optimal solution for payments when they are contingent on output; any structure where payments for higher realisations are sufficiently large is optimal. In this section, it is demonstrated that under uncertainty, the optimal solution with respect to payments is unique, taking the form of a linear contract where landowners receive a fixed share of the value of output. Again, the uncertainty set is defined by Equation (4.18), with the additional assumptions described at the start of Section 4.5.2. Whereas previously only ES output was uncertain, if payments are output-based, the landowner must additionally contend with uncertainty. The objective function for the robust contract design problem with moral hazard and output-based payments is:

$$\max_{x,\mathbf{F}} \min_{\mathbf{a}} \{\mathbb{E}_{\phi}[v(x) - C(x, e_2) - \lambda F]\} \tag{4.49}$$

The objective function is still subject to the $(n+1)$ IR_j constraints in Equation (4.43), and the IR_0 constraint remains binding ($F_0^{O,R,MH} = C(x^{O,R,MH}, e_2)$). The superscript O, R, MH describes the optimal robust output-based solution with moral hazard. Uncertainty alters the *robust* $MHIC$ constraint. Incorporating uncertainty over the probability distribution $\mathbf{a}$, the robust $MHIC$ is given by:

$$\min_{\mathbf{a}} \sum_{j=1}^{n} a_j e_2 F_j + (1 - \sum_{j=1}^{n} a_j e_2) F_0 - C(x, e_2) \geq \min_{\mathbf{a}} \sum_{j=1}^{n} a_j e_1 F_j + (1 - \sum_{j=1}^{n} a_j e_1) F_0 - C(x, e_1) \tag{4.50}$$

The robust $MHIC$ introduces a new feature to the problem compared to previous sections. Landowner payments are also subject to uncertainty. The principal designs the payment scheme to ensure incentive compatibility from that in the worst-case, hence the term $\min_{\mathbf{a}}$. To simplify, we

rearrange terms and recognize that the left-hand side must exceed the right-hand side by a margin at least equal to the cost difference $\psi(x, e_2) = C(x, e_2) - C(x, e_1)$ associated with higher effort. This leads to:

$$\min_{\mathbf{a}} \sum_{j=1}^{n} a_j(e_2 - e_1)w_j(x) \geq \psi(x, e_2) \tag{4.51}$$

With output-based payments, uncertainty also becomes a part of the landowner problem. The $MHIC$ is binding, and the term $\min_{\mathbf{a}} \sum_{j=1}^{n} a_j(e_2 - e_1)w_j$ can be expanded to produce $\sum_{j=1}^{n} \bar{a}_j w_j(x) - \sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}$[5]. Equation (4.50) can be reformulated as:

$$\sum_{j=1}^{n} \bar{a}_j w_j(x) = \frac{\psi(x, e_2)}{(e_2 - e_1)} + \sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)} \tag{4.52}$$

Where the uncertainty adjustment for the landowner has been rearranged. Interestingly, while the principal responds to uncertainty by incorporating a negative adjustment to social welfare from maximisation, the landowner responds to uncertainty by requiring a higher payment to be incentivised into compliance. Exposure to uncertainty results in the landowner requiring a premium to be induced into compliance, represented by the term $\sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}$. Substituting the term $\frac{\psi(x,e_2)}{(e_2-e_1)} + \sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}$ on the right-hand side of Equation (4.52) into the principal's objective function provides the following robust optimisation problem with respect to x and $\mathbf{w}(\mathbf{x})$:

[5]Recall from Ben-Tal and Nemirovski (1999) that $\min_{a \in \mathcal{U}} \langle \mathbf{a}, \mathbf{x} \rangle = \sum_j \bar{a}_j x_j - \sqrt{\sum_j \delta_j^2 x_j^2}$

$$\max_{x,\mathbf{w}(\mathbf{x})} \sum_{j=1}^{n} \bar{a}_j e_2(\phi_j - \phi_0)x - \lambda e_2 \left(\frac{\psi(x, e_2)}{(e_2 - e_1)} + \sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)} \right) + \phi_0 x - (1+\lambda)C(x, e_2)$$
$$-e_2 \sqrt{\sum_{j=1}^{n} \delta_j^2 \Big((\phi_j - \phi_0)x - \lambda w_j(x) \Big)^2} \tag{4.53}$$

The derivation of Equation (4.53) can be found in Section 4.7.6 of the Appendix. The optimal payment when ϕ_j is realised can be derived from the first-order conditions with respect to $w_j(x)$:

$$\frac{\delta_j^2 \Big((\phi_j - \phi_0)x - \lambda w_j(x) \Big)}{\sqrt{\sum_{j=1}^{n} \delta_j^2 \Big((\phi_j - \phi_0)x - \lambda w_j(x) \Big)^2}} - \frac{\delta_j^2 w_j(x)}{\sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}} = 0 \tag{4.54}$$

Notice that Equation (4.54) requires that $(\phi_j - \phi_0)x - \lambda w_j(x) > 0 \forall j$; if this is not true the optimisation problem has no solution. This condition then only holds when:

$$\frac{w_j(x)}{(\phi_j - \phi_0)x - w_j(x)} = \beta, \beta \in [0, \infty) \tag{4.55}$$

This implies that an optimal contract is described by $w_j(x) = \alpha(\phi_j - \phi_0)x, \forall \alpha \in [0, 1]$[6]. While Equation (4.55) determines that any payment structure such that the landowner receives share α of the value of ES output above the baseline is sufficient, the optimal share α^*, is defined as the proportion of output necessary for the $MHIC$ constraint in Equation (4.50) to hold with equality:

[6]In Section 4.7.7 of the Appendix, we derive and show how this contract is optimal relative to the first-order conditions described by Equation (4.54). Yu and Kong (2020) further show that this type of simple linear contract is uniquely optimal.

$$\alpha^* = \frac{\psi(x)}{(e_2 - e_1)x\Big(\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) - \sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}\Big)} \tag{4.56}$$

Equation (4.56) shows that the optimal share of the value of output to induce the landowner into compliance is a function of the land input allocation, x. Whether α^* is increasing in x will depend on the functional form of input costs. If $\phi(x)$ takes a linear form, e.g., ax with $a > 0$, α^* is always increasing in x (for a more in-depth discussion on this see Section 4.7.8 of the Appendix). We can substitute $w_j(x) = \alpha^*(x)(\phi_j - \phi_0)x$ into Equation (4.53) to solve for the optimal land input allocation:

$$\max_x \sum_{j=1}^{n} \bar{a}_j e_2\Big((1 - \lambda\alpha^*(x))(\phi_j - \phi_0)x\Big) + \phi_0 x - (1 + \lambda)C(x, e_2) - e_2\sqrt{\sum_{j=1}^{n} \delta_j^2\Big((1 - \lambda\alpha^*(x))(\phi_j - \phi_0)x\Big)^2} \tag{4.57}$$

Solving Equation (4.57) with respect to x leads to Proposition 4:

Proposition 4 *Given uncertainty, when effort is not verifiable but output is measurable, a hybrid payment structure is again optimal, with a base payment covering landowner conservation costs. The optimal bonus structure involves the principals sharing a fraction of the value of output with the landowner. The landowner requires an uncertainty premium to compensate for the risks associated with a little-understood relationship between effort and output. The impact this has on land allocation depends on the characteristics of the optimal share of the value of output required to induce effort by the landowner and the cost of exerting effort.*

As discussed in Proposition 4, with moral hazard and uncertainty, a linear bonus structure that increases in output is optimal given the use of output-based payments. The contract includes a base payment of $C(x^{O,R,MH}, e_2)$, and bonus payments $\mathbf{w}(\mathbf{x}) = C(x^{O,R,MH}, e_2) - \alpha^*(x^{O,R,MH})\phi_0 x^{O,R,MH}$ so that participation is always incentivised. The principal provides the landowner with a fixed share

of the value of output to incentivise compliance. Further, the solution for the optimal land input allocation is:

$$c'(x^{O,R,MH}) = \frac{1}{1+\lambda}\left[\sum_{j=1}^{n}\bar{a}_j e_2(\phi_j-\phi_0)\left(1+\lambda(x^{O,R,MH}\alpha'(x^{O,R,MH})-\alpha^*(x^{O,R,MH}))\right)+\phi_0 \right.$$
$$\left. -e_2\cdot\frac{\sum_{j=1}^{n}\delta_j^2(\phi_j-\phi_0)x^{O,R,MH}\Big(1+\lambda(x^{O,R,MH}\alpha'(x^{O,R,MH})-\alpha^*(x))\Big)}{\sqrt{\sum_{j=1}^{n}\delta_j^2\Big((1-\lambda\alpha^*(x^{O,R,MH}))(\phi_j-\phi_0)x^{O,R,MH}\Big)^2}}\right]-\psi'(x^{O,R,MH}) \tag{4.58}$$

Whether the $x^{O,R,MH}$ derived by Equation (4.58) is lesser or greater than those derived in preceding sections depends on the relationship between x and $\alpha^*(x)$. To illustrate, we describe in Section 8.9 of the Appendix the impact of uncertainty on land input when $\psi(x)$ is linear in x. Briefly, the principal faces a trade-off. By compensating the landowner with a fixed share of the value of output, the principal also reduces social welfare gains from conservation, which results in reduced land input.

4.5.5 Discussion

There are a few major policy takeaways from this research endeavour. Firstly, our findings show that under uncertainty, the robust contract involves a more conservative approach, reducing investments in ES to shield against worst-case outcomes. While these contracts ensure a performance guarantee, they can also result in lower expected social welfare relative to a standard contract. These results align with considerations regarding uncertainty that have also been made by practitioners. For example, Costa Rica proposed that Reducing Emissions from Deforestation and Forest Degradation (REDD+) payments follow a 'conservativeness' principle, compensating nations based on the lower bound of a 95% confidence interval for carbon stock changes or emissions reductions (Angelsen 2008). This approach mirrors the conservative behavior prescribed by our

model, where the principal minimises exposure to uncertainty by reducing investment into ES.

Secondly, with moral hazard, the optimal contract is never a pure output-based program, but a hybrid scheme where landowners are provided a base payment to induce participation and incentivised to exert effort with bonus payments increasing in output. Table 4.3 summarises our findings in the previous sections regarding the structure of optimal contracts given moral hazard and uncertainty. The vast majority of PES schemes remain input-based. However, it is unclear how successful PES schemes are in achieving their environmental objectives, and there is increasing interest in the potential for PES with output-based payments to improve conditionality. We have shown that where output is observable, providing the landowner with a fixed share of the value of output is sufficient to share risk and induce compliance without the need to monitor.

While output-based programs are new, there is now an empirical literature that discusses other benefits of the approach. It also encourages landowners to innovate, and elevates ES production from an obligation to an integral aspect of farming (Herzon et al. 2018). However, ES output is often not observable. Some schemes assign an intermediate proxy for output (e.g., Wezel et al. 2016), but this may not always be possible (Wunder et al. 2020). Our findings describe a potential factor in why the vast majority of PES schemes use input-based payments. If tying compensation to outcomes is the best way to ensure a performance guarantee, the challenge of measuring ES output accurately limits the feasibility of such schemes. Input-based payments remain prevalent as they are easier to implement and monitor, despite their potential inefficiencies. Another approach is to make payments contingent on *modelled output*. Here, payments are assigned to landowners based on the estimated benefits using decision tools (e.g., Fales et al. 2016). However, such schemes are exposed to uncertainty. Payment levels again correspond to the actions landowners take, which in turn reflect model predictions that are not directly observable. Thus, such schemes are largely dependent on the certainty of models (Bartkowski et al. 2021). While they may improve the cost-effectiveness of PES programs (Sidemo-Holm et al. 2018), our findings show that such schemes may not be optimal where uncertainty is very large.

Table 4.3: Optimal Payment Structures under Different Scenarios with Moral Hazard

Scenario	Optimal Contract
Effort verifiable, but costly and/or output non-verifiable. ES production known with certainty.	Optimal payment compensates landowners up to conservation costs. Principal ensures input-conditionality (exertion of effort) by monitoring and penalising non-compliance (shirking). Monitoring result in a downward distortion in land input, as the principal seeks to balance the benefits of increased effort with the costs imposed by verifying compliance. (**Ozanne et al. 2001; White and Hanley 2016**)
Effort verifiable, but costly and/or output non-verifiable. ES production not known with certainty.	Optimal payment still compensates landowner up to their conservation costs and induces compliance with monitoring and penalties resulting in a distortion to land input. There is an additional downward pressure induced by uncertainty, as landowners reduce investment into ES provision to improve their payoff in their worst-case scenario. (**Proposition 2**)
ES output verifiable and/or effort non-verifiable. ES production known with certainty.	Hybrid payment structure optimal. Landowners are compensated up to their conservation costs regardless of output as a base payment. There are additionally bonus payments that increase in output to ensure landowners exert effort, any variable payment structure is optimal as long as the expected value of bonuses exceeds the expected cost of exerting effort. The principal thus motivates the landowner to exert effort by providing higher payments when higher output is realized. (**Proposition 3**)
ES output verifiable and/or effort non-verifiable. ES production not known with certainty.	Again a hybrid payment structure is optimal. However, while without uncertainty, there is no unique solution to the optimal structure of contracts, given uncertainty, the unique optimal payment structure involves the principals sharing a fixed share of the value of output with the landowner as a bonus payment. The landowner requires an uncertainty premium to compensate for the risks associated with a little-understood relationship between effort and output. The impact this has on land allocation is indeterminate. (**Proposition 4**)

4.6 Conclusion

In this study, we extend the standard principal-agent framework in an agri-environmental context to describe the implications of uncertainty on PES contracting. We show that, given the standard input-based contract, an uncertainty-averse principal can shield themselves from worst-case scenarios regarding ES provision by adopting max-min preferences which ensure a performance guarantee even under the most unfavorable conditions. In our first model, this uncertainty has no impact on landowner participation, as they need only be compensated for their conservation costs. These findings are validated by a numerical exercise and simulation using data on soybean and grain corn cultivation in Ontario, in addition to the findings of a pan-Canadian watershed ES valuation. In the simulation, we also show the trade-off between robustness and contract performance. A principal protecting themselves against worst-case outcomes ultimately lowers expected social welfare given the realisation of other potential scenarios in the uncertainty set.

Next, we introduce another degree of complexity to the problem by considering moral hazard, an obstacle in real-world PES schemes that makes it difficult to ensure conditionality. Landowners are contracted for two inputs: land and effort. If payments are input-based, there is little change to the structure of the robust scheme, again showing that a conservative approach to ES investment is optimal to adjust to uncertainty. Uncertainty has greater implications for the structure of output-based contracts. Basing payments on output allows principals to offload some of the negative consequences of uncertainty onto the landowner. In the optimal scheme, the principal offers landowners a hybrid payment, offering a base payment to induce participation, and a fixed share of the value of ES output as a bonus to induce compliance. However, this risk-sharing measure does not unambigiously protect the principal from uncertainty, although the impact on land input is uncertain.

This study is not without limitations. The numerical example and simulation are limited, particularly due to the lack of available data on the costs of monitoring landowner participation in agri-environmental schemes in Canada. If suitable data were collected, a more expansive simulation exercise could describe the consequences of uncertainty with moral hazard on social welfare.

We assume that landowners are largely insulated from environmental risk, i.e., their conservation costs are known with certainty, and that they are risk-neutral. If conservation costs include foregone agricultural profits, the first of these assumptions is too strong. Future work can extend our analysis by introducing more complexity to the landowner in this respect. We also simplify the principal problem by assuming that it is possible to set a monitoring so that the landowner always participates and complies. An extension could be to incorporate many landowners, and the potential for some landowners to shirk at the optimum while others do not (e.g., Hart and Latacz-Lohmann 2005). Another major limitation is our treatment of time. PES schemes are not one-time contracts, but can span many years, with renegotiation and changing dynamics between landowners and ecosystems (Hayes et al. 2022). Further, incorporating time would also have great implications for our treatment of uncertainty. Some PES schemes also set up programs to begin monitoring environmental objectives, with the goal of establishing baselines and improving know-how (e.g., Corbera et al. 2007). Uncertainty, and how PES interacts with our knowledge of ES have a huge impact on our understanding.

Nevertheless, our research represents a step forward in the study of PES contracting, and has important implications for policymakers. While the number of PES schemes continues grow, the relationships underlying land use and ES often remain obfuscated. Further, policymakers are also increasingly interested in novel techniques to ensure conditionality, that buyers receive the services they are paying for. We have shown that uncertainty surrounding ES production has important implications for PES and cannot be ignored, but instead should be considered in contract design to ensure that schemes are successful in achieving the environmental objectives desired.

4.7 Appendix

4.7.1 Derivation of Equation (4.21)

This result is derived from Theorem 3.1 in Ben-Tal and Nemirovski (1999). They show that, given an ellipsoidal structure of the uncertainty set with the form:

$$\mathcal{U} = \{(\bar{a}_1, ..., \bar{a}_n) + (\delta_1 u_1, ..., \delta_n u_n) | ||\mathbf{u}|| \leq 1\} \tag{4.59}$$

We have that:

$$\min_u \sum_j a_j x_j = \min_u \sum_j (\bar{a}_j + \delta_j u_j) x_j = \sum_j \bar{a}_j x_j - \max_u \sum_j \delta_j u_j x_j \tag{4.60}$$

Now note that, by the Cauchy-Schwarz inequality, the term $\max_u \sum_j \delta_j u_j x_j$ is $\sqrt{\sum_j \delta_j^2 x_j^2}$ (Yu and Kong 2020). Thus, Equation (4.60) is:

$$\min_u \sum_j a_j x = \sum_j \bar{a}_j x - \sqrt{\sum_j \delta_j^2 x_j^2} \tag{4.61}$$

4.7.2 Estimation of ES value

Note that the ES value in the numerical example is derived using an OLS specification, where the regression coefficients are shown in Table 4.4 below.

Table 4.4: Structural Equation Model Regression Results

Variables	(1) Costs		(2) Turbidity	
Turbidity	0.047*	(0.022)		
Forest	-0.618	(0.403)	-0.671*	(0.339)
Agriculture	-0.363	(0.451)	1.617***	(0.429)
Urban	-0.511	(0.371)	1.750***	(0.359)
Wildfire	109.65	(384.63)	302.565	(701.36)
Population	-0.490***	(0.049)		
Seasonal	0.055*	(0.026)		
Groundwater	0.001	(0.004)		
Residential	-0.000	(0.002)		
Losses	0.001	(0.003)		
ICI	0.001	(0.004)		
Days $> 90\%$	0.043	(0.038)		
Filtration	0.681***	(0.186)		
Coagulation	0.147	(0.116)		
Pre-treatment	-0.007	(0.084)		
Electricity	0.174	(0.382)		
Labour	1.883*	(0.968)		
Constant	0.606	(4.305)	1.485***	(0.222)
No. observations	377		406	
Adjusted R^2	0.647		0.182	
F-statistic	41.48		23.49	

[1] Note: * $p < 0.1$, ** $p < 0.05$, *** $p < 0.01$.

In the first-stage regression, the variable of interest is taken as the difference between the coefficients for cropland and forest, $\beta_{crop} - \beta_{for}$. As variance-covariance matrices for confidential datasets by Statistics Canada are not releasable, we assume the covariance between β_{crop} and β_{for} is zero, and the variance of this term is taken as $Var(\beta_{crop}) + Var(\beta_{for})$.

Finally, to construct a 90% confidence interval, we use the Sobel test statistic on th indirect effect, which is a product of the term described above and the coefficient for turbidity.

4.7.3 Crop density of soybeans and corn in Ontario

Land Type	Soybean	Grain corn
1	120.57	64.88
2	173.32	108.50
3	38.14	29.48
4	68.25	84.08
5	11.80	10.93
6	20.18	15.99
7	8.36	15.99
8	7.27	5.46
9	8.24	6.38

Table 4.5: Soybean and Corn Densities by Land Type

4.7.4 Lemma $4.5.1$

Equation (4.34) can be expanded and rearranged to produce:

$$p\eta \geq C(x, e_2) - C(x, e_1) = c(x) + \psi(x, e_2) - (c(x) + \psi(x, e_1)) \tag{4.62}$$

Cancelling like terms here will give us Equation (4.35), but as an inequality. To show this is binding, we need only to consider that in the social welfare function, monitoring is associated with a cost. Now, suppose that the $MHIC$ was not binding. Then $p - \frac{\Delta\psi(x)}{\eta} > 0$. Consequently, note that p could be reduced by some $\epsilon > 0$ without violating $MHIC$, ensuring that the landowner still complies. Further, the decrease in the monitoring rate would increase social welfare by $(1+\lambda)\Big(M(\frac{\Delta\psi(x)}{\eta}+\epsilon) - M(\frac{\Delta\psi(x)}{\eta})\Big)$. The principal has an incentive to minimise payments to the landowner, but only subject to the $MHIC$ constraints that guarantee landowner compliance. Therefore, at the optimum, the principal will set the monitoring rate such that Equation (4.35) is binding.

4.7.5 Derivation of Equation (4.46)

Starting with the moral hazard incentive compatibility ($MHIC$) constraint in Equation (4.44):

$$\sum_{j=1} a_j e_2 F_j + \left(1 - \sum_{j=1} a_j e_2\right) F_0 - C(x, e_2) \geq \sum_{j=1} a_j e_1 F_j + \left(1 - \sum_{j=1} a_j e_1\right) F_0 - C(x, e_1) \tag{4.63}$$

This constraint ensures that the expected payoff for the landowner is greater when exerting high effort e_2 than when exerting low effort e_1. Expanding each term we get:

$$\sum_{j=1} a_j e_2 (F_j - F_0) - C(x, e_2) \geq \sum_{j=1} a_j e_1 (F_j - F_0) - C(x, e_1) \tag{4.64}$$

Where the baseline payment F_0 cancels out. This can be rearranged to produce:

$$\sum_{j=1} a_j (e_2 - e_1) F_j - \sum_{j=1} a_j (e_2 - e_1) F_0 \geq C(x, e_2) - C(x, e_1) \tag{4.65}$$

Now, recall from Equation (4.31) that the cost difference between exerting high and low effort, $\Delta C(x)$, can be expressed as:

$$C(x, e_2) - C(x, e_1) = \psi(x), \tag{4.66}$$

where $\psi(x)$ represents the additional cost associated with the higher effort level e_2. Substituting this into the inequality gives:

$$\sum_{j=1} a_j (e_2 - e_1)(F_j - F_0) \geq \psi(x) \tag{4.67}$$

Dividing both sides by $(e_2 - e_1)$ yields:

$$\sum_{j=1} a_j(F_j - F_0) \geq \frac{\psi(x)}{(e_2 - e_1)} \tag{4.68}$$

Finally, by defining $w_j(x) = F_j - F_0$, we obtain the simplified form:

$$\sum_{j=1}^{n} a_j w_j(x) \geq \frac{\psi(x)}{(e_2 - e_1)} \tag{4.69}$$

4.7.6 Derivation of Equation (4.53)

We start from the following objective function as described by Equation (4.49):

$$\max_{x,\mathbf{F}\geq 0} \min_{\mathbf{a}} \mathbb{E}_\phi \left[v(x) - C(x, e_2) - \lambda \mathbf{F} \right]. \tag{4.70}$$

We can expand $\mathbb{E}_\phi$ to produce the following:

$$\max_{x,\mathbf{F}\geq 0} \min_{\mathbf{a}} \{ \sum_{j=1} \bar{a}_j e_2 \phi_j x + (1 - \sum_{j=1} \bar{a}_j) e_2 \phi_0 x - \lambda \Big(\sum_{j=1} \bar{a}_j e_2 F_j + (1 - \sum_{j=1} \bar{a}_j) e_2 F_0 \Big) \} - C(x, e_2) \tag{4.71}$$

By rearranging terms, and noting again that the baseline terms $\phi_0 x$ and F_0 are constant, we can simplify Equation (4.71) to produce:

$$\max_{x,\mathbf{F}\geq 0} \min_{\mathbf{a}} \{ \sum_{j=1}^{n} \bar{a}_j e_2 (\phi_j - \phi_0) x - \lambda (F_j - F_0) \} + \phi_0 x - C(x, e_2) - \lambda F_0 \tag{4.72}$$

The IR_0 constraint implies that $F_0 = C(x, e_2)$. Thus, this can be simplified to produce:

$$\max_{x,\mathbf{F}\geq 0} \min_{\mathbf{a}} \{\sum_{j=1}^{n} \bar{a}_j e_2(\phi_j - \phi_0)x - \lambda(F_j - F_0)\} + \phi_0 x - (1+\lambda)C(x, e_2) \tag{4.73}$$

We can again rewrite $F_j - F_0 = w_j(x)$. Substituting this into the problem produces:

$$\max_{x,\mathbf{w}(\mathbf{x})\geq 0} \min_{\mathbf{a}} \{\sum_{j=1} \bar{a}_j e_2(\phi_j - \phi_0)x - \lambda w_j(x)\} + \phi_0 x - (1+\lambda)C(x, e_2) \tag{4.74}$$

Finally, we again use Ben-Tal and Nemirovski (1999) to expand the term $\min_{\mathbf{a}} \sum_{j=1}^{n} \bar{a}_j e_2(\phi_j - \phi_0)x - \lambda w_j(x)$ into $\sum_{j=1}^{n} \bar{a}_j e_2\Big((\phi_j - \phi_0)x - \lambda w_j(x)\Big) - \sqrt{\sum_{j=1}^{n} \delta_j^2 e_2^2\Big((\phi_j - \phi_0)x - \lambda w_j(x)\Big)^2}$. Substituting this into Equation (4.74) produces:

$$\max_{x,\mathbf{w}(\mathbf{x})\geq 0} \sum_{j=1}^{n} \bar{a}_j e_2(\phi_j - \phi_0)x - \sum_{j=1}^{n} \bar{a}_j e_2 \lambda w_j(x) - e_2\sqrt{\sum_{j=1}^{n} \delta_j^2\Big((\phi_j - \phi_0)x - \lambda w_j(x)\Big)^2} + \phi_0 x - (1+\lambda)C(x, e_2) \tag{4.75}$$

By Equation (4.52), the $MHIC$ requires that $\sum_{j=1}^{n} \bar{a}_j w_j(x) = \frac{\psi(x,e_2)}{(e_2 - e_2)} + \sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}$. We substitute this expression into Equation (4.75) to produce:

$$\begin{aligned} \max_{x,\mathbf{w}(\mathbf{x})\geq 0} & \sum_{j=1}^{n} \bar{a}_j e_2(\phi_j - \phi_0)x - e_2\lambda\Big(\frac{\psi(x, e_2)}{(e_2 - e_2)} + \sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}\Big) \\ & - e_2\sqrt{\sum_{j=1}^{n} \delta_j^2\Big((\phi_j - \phi_0)x - \lambda w_j(x)\Big)^2} + \phi_0 x - (1+\lambda)C(x, e_2) \end{aligned} \tag{4.76}$$

4.7.7 Derivation of Equation (4.55)

Equation (4.54) is derived by taking the derivative of Equation (4.53) with respect to any $w_j(x)$. Equation (4.54) can be rearranged to produce:

$$\frac{w_j(x)}{(\phi_j - \phi_0)x - w_j(x)} = \frac{\sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}}{\sqrt{\sum_{j=1}^{n} \delta_j^2 \Big((\phi_j - \phi_0)x - w_j(x)\Big)^2}}, \tag{4.77}$$

Since $\frac{\sqrt{\sum_{j=1}^{n} \delta_j^2 w_j^2(x)}}{\sqrt{\sum_{j=1}^{n} \delta_j^2 \Big((\phi_j - \phi_0)x - w_j(x)\Big)^2}}$ is constant with respect to each individual $w_j(x)$ (as it depends only on the aggregate values across all j), we define this ratio as β:

$$\frac{w_j(x)}{(\phi_j - \phi_0)x - w_j(x)} = \beta \tag{4.78}$$

As the terms in the numerator and denominator are both positive, it must be the case that $\beta \in (0, \infty)$. Further, we can derive the optimal form solely in terms of $w_j(x)$ by solving Equation (4.78) in terms of $(\phi_j - \phi_0)x$. This expression can be rearranged in terms of $w_j(x)$:

$$w_j(x) = \beta\Big((\phi_j - \phi_0)x - w_j(x)\Big). \tag{4.79}$$

Expanding and rearranging terms, we get:

$$w_j(x) + \beta w_j(x) = \beta(\phi_j - \phi_0)x. \tag{4.80}$$

This simplifies to:

$$w_j(x) = \frac{\beta}{1+\beta}(\phi_j - \phi_0)x. \tag{4.81}$$

To make this expression more interpretable, we define a new parameter α as follows:

$$\alpha = \frac{\beta}{1+\beta}. \tag{4.82}$$

With this definition, we can express $w_j(x)$ in the simplified form:

$$w_j(x) = \alpha(\phi_j - \phi_0)x, \tag{4.83}$$

where $\alpha \in (0, 1)$ because $\beta \in [0, \infty)$.

To show this is optimal, we substitute $w_j(x) = \alpha(\phi_j - \phi_0)x$ into Equation (4.54):

$$\frac{\delta_j^2\Big((\phi_j - \phi_0)x - \lambda w_j(x)\Big)}{\sqrt{\sum_{j=1}^n \delta_j^2\Big((\phi_j - \phi_0)x - \lambda w_j(x)\Big)^2}} - \frac{\delta_j^2 w_j(x)}{\sqrt{\sum_{j=1}^n \delta_j^2 w_j^2(x)}} \tag{4.84}$$

Substituting $w_j(x) = \alpha(\phi_j - \phi_0)x$ into both terms, we get:

$$= \frac{\delta_j^2\Big((\phi_j - \phi_0)x - \lambda\alpha(\phi_j - \phi_0)x\Big)}{\sqrt{\sum_{j=1}^n \delta_j^2\Big((\phi_j - \phi_0)x - \lambda\alpha(\phi_j - \phi_0)x\Big)^2}} - \frac{\delta_j^2\alpha(\phi_j - \phi_0)x}{\sqrt{\sum_{j=1}^n \delta_j^2\Big(\alpha(\phi_j - \phi_0)x\Big)^2}} \tag{4.85}$$

Next, factor out $(\phi_j - \phi_0)x$ in the numerators and denominators:

$$= \frac{\delta_j^2(\phi_j - \phi_0)x(1-\lambda\alpha)}{\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2x^2(1-\lambda\alpha)^2}} - \frac{\delta_j^2\alpha(\phi_j-\phi_0)x}{\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2x^2\alpha^2}} \tag{4.86}$$

Then, cancel $(\phi_j - \phi_0)x$ from each fraction:

$$= \frac{\delta_j^2(1-\lambda\alpha)}{\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2(1-\lambda\alpha)^2}} - \frac{\delta_j^2\alpha}{\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2\alpha^2}} \tag{4.87}$$

Taking $(1 - \lambda\alpha)$ and α out of the square roots in the denominators:

$$= \frac{\delta_j^2(1-\lambda\alpha)}{|1-\lambda\alpha|\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2}} - \frac{\delta_j^2\alpha}{|\alpha|\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2}} \tag{4.88}$$

Finally, combine terms by factoring out $\frac{\delta_j^2}{\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2}}$:

$$= \frac{\delta_j^2}{\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2}} \left(\frac{1-\lambda\alpha}{|1-\lambda\alpha|} - \frac{\alpha}{|\alpha|} \right) \tag{4.89}$$

Thus, the simplified form is:

$$\frac{\delta_j^2}{\sqrt{\sum_{j=1}^n \delta_j^2(\phi_j-\phi_0)^2}} \left(\frac{1-\lambda\alpha}{|1-\lambda\alpha|} - \frac{\alpha}{|\alpha|} \right) = 0 \tag{4.90}$$

Note that this is true for *any* $\alpha \in (0, 1)$.

4.7.8 Responsiveness of $\alpha^*(x)$ to changes in x

By the quotient rule, we have that:

$$\frac{d\alpha^*(x)}{dx} = \frac{v(x)u^{'}(x) - u(x)v^{'}(x)}{[v(x)]^2} \tag{4.91}$$

Where $u(x) = \psi(x)$, $v(x) = (e_2 - e_1)x\Big(\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) - \sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}\Big)$. Therefore, we can state that:

$$\frac{d\alpha^*}{dx} = \frac{x\psi^{'}(x) - \psi(x)}{(e_2 - e_1)x^2\Big(\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) - \sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)}\Big)} \tag{4.92}$$

The exact relationship between the optimal payment share and x depends on the cost function. For the case where $\psi(x) = ax, a > 0$, Equation (4.92) becomes:

$$\alpha^* = \frac{a}{(e_2 - e_1)\Big(\sum_{j=1}^{n} \bar{a}_j(\phi_j - \phi_0) - \sqrt{\sum_{j=1}^{n} \delta_j^2(\phi_j - \phi_0)^2}\Big)} \tag{4.93}$$

α^* is increasing in a, indicating that, as the cost of conservation rises, so too does the share of the value of output required to incentivise the landowner to exert effort.

4.7.9 Relationship between α^* and land input when $\psi(x)$ is linear in x

Let $\psi(x)$ be linear in x. Then, given Equation (4.56), clearly $\alpha^{'}(x) = 0$:

We can use this to simplify Equation (4.58) in Section 6.4 as:

$$c'(x^{O,R,MH}) = \frac{1}{1+\lambda}\Bigg(\sum_{j=1}^{n}\bar{a}_j e_2(\phi_j-\phi_0)(1+\lambda(-\alpha^*))+\phi_0 \\ - e_2\cdot\frac{\sum_{j=1}^{n}\delta_j^2(\phi_j-\phi_0)x^{O,R,MH}\,(1+\lambda(-\alpha^*))}{\sqrt{\sum_{j=1}^{n}\delta_j^2\Big((1-\lambda\alpha^*)(\phi_j-\phi_0)x^{O,R,MH}\Big)^2}}\Bigg)-\psi'(x^{O,R,MH}) \tag{4.94}$$

The third term in this expression can be simplified by bringing expanding out square terms in the denominator to produce:

$$c'(x^{O,R,MH}) = \frac{\sum_{j=1}^{n}{}^{n}\bar{a}_j e_2(\phi_j-\phi_0)(1-\lambda\alpha^*)+\phi_0}{(1+\lambda)} - \frac{e_2\cdot\sum_{j=1}^{n}{}^{n}\delta_j^2(\phi_j-\phi_0)}{(1+\lambda)\sqrt{\sum_{j=1}^{n}{}^{n}\delta_j^2(\phi_j-\phi_0)^2}}-\psi'(x^{O,R,MH}) \tag{4.95}$$

In this instance, observe that uncertainty continues to contribute to a downward distortion in the contracted land input. However, the nature of this distortion changes significantly. Now, the principal faces a trade-off. Compensating the landowner with a fixed share of the value of output induces them to exert effort, which can offload some of the negative consequences of uncertainty to the principal. However, this also reduces social welfare gains from conservation, hence the term $(1-\lambda\alpha^*)$ in the first term in Equation (4.95).

Chapter 5

Conclusion

This thesis has explored the challenges of uncertainty and conditionality to Payments for Watershed Services (PWS). It comprises three research papers that contribute to our knowledge of the optimal provision of ecosystem services (ES) through novel and rigorous analysis. While each chapter has addressed distinct aspects of PWS, they are interlinked, and all three provide new insights into improving the performance of conservation schemes.

The second chapter addressed: **What do we know about forested watershed ES and the determinants of their value**? While few PWS schemes conduct a formal valuation exercise prior to implementation (Porras 2008), they can serve as a useful complementary step in clarifying the potential economic benefits of conservation. There have been many meta-analyses on the valuation of forest ES in various contexts (e.g., Zandersen and Tol 2005; Lindhjem 2007; Ojea and Martin-Ortega 2015). This study contributes to our knowledge by conducting the first meta-analysis of the global forested watershed ES valuation literature. Using a meta-regression approach in conjunction with the Common International Classification of Ecosystem Services (CICES) framework to distinguish between different services, the results identified several commonalities and trends in the literature. Firstly, the findings highlighted that the classification of ecosystem services are a significant factor in explaining variation in estimates. This is an important outcome because it demonstrates that the specific good or service under valuation is a key driver for its market and

non-market value. The erosion control function of forests was consistently found to provide higher benefits to society. Further, increasing precision within the CICES framework greatly increased the explanatory and predictive power of the model. The study also identified other key determinants of value such as types of end-users and valuation methodology. Secondly, the model also refined the existing literature by testing the sensitivity to scale of ES by including either watershed size or forest size as explanatory variables. The existing literature typically uses national forest area as a proxy for the supply of forest ES. This may overestimate the impact on the marginal value of increasing size and possible decreasing marginal utility.

The third chapter asked: **How can we use valuation to extend our knowledge and reduce uncertainty surrounding the value of watershed ES**? The focus of this section shifted further to the supply side of PWS, examining how forest cover and wildfires influence the cost of drinking water treatment in Canada. Drinking water treatment facilities are often buyers in real-world PWS schemes (e.g., Pagiola 2008; Huang et al. 2009). Determining the value provided to communities by forested watersheds can guide their willingness-to-invest into PWS and inform policy decisions on source water protection. While similar valuation exercises have been carried out in other contexts (Abildtrup et al. 2013; Vincent et al. 2016; Warziniack et al. 2017), this is the first study of its kind in Canada. Further, this is the first study to incorporate wildfire disturbance as a determinant of the costs of drinking water treatment in the literature. Data on drinking water treatment processes was coupled with surrounding land use to determine the savings generated by conservation. The results demonstrated a clear economic rationale for forest conservation. The models found a strong, statistically significant relationship between increased forested land, particularly in closer vicinity to treatment facilities, and lower treatment costs. However, the analysis also uncovered an even stronger relationship between the emerging threat of wildfires and treatment costs. Wildfire disturbances, even or great distances away from a facility, drastically degrade raw water quality, and this in turn increases treatment costs. The key policy takeaway of this chapter is to illustrate the trade-off between the benefits of forested land and the risk posed to source water by the increasing likelihood of wildfires. In the Pacific Northwest, nearly three-quarters of water

utilities are at moderate risk of wildfire with insufficient risk management strategies (Robinne et al. 2019). In Canada, where climate change is anticipated to increase the number of days where wildfires are likely, as well as the number of days where fire intensities exceed current suppression resources (Wotton et al. 2017), demonstrating the economic impact of wildfires on source water is paramount to justify such costly risk management strategies.

The fourth chapter asked: **Given persistent uncertainty, how can we design PES contracts that ensure better outcomes**? The focus here is on design of PES schemes, addressing how the design of contracts can ensure the participation and compliance of landowners under uncertainty. This study extended the standard principal-agent framework employed by past research (e.g., Ozanne and White 2007), and employed novel robust optimisation techniques to describe how a principal can obtain a performance guarantee (i.e., ensure the best worst-case performance) when they are uncertain about the relationship between conservation and the provision of ES. In the model, a principal induces a landowner to produce ES through conservation in return for compensation. However, the principal and landowner lack precise knowledge on the parameters that govern output, only knowing they belong within the range of some uncertainty set. The results indicated that when compensation is input-based, and inputs are easily verifiable, the only way for a principal to secure a performance guarantee is to invest conservatively in ES. This result holds also with the inclusion of adverse selection and moral hazard in this context. However, when input is not verifiable and payments are output-based, the principal can ensure better worst-case outcomes by providing the landowner with a bonus payment structure that is directly tied to output. The poor performance of some PES schemes has led to a discussion on the potential for output-based schemes to ensure conditionality by tying compensation to outcomes. As interest in output-based PWS grows, there are two approaches to tying compensation to output: schemes where payments are contingent on *output indicators*, and those where they are tied to *modelled output*. With uncertainty, output-based schemes work best when output is observable to both the principal and landowners; however, this is unfortunately rarely possible due to challenges related to ES measurement (Bartkowski et al. 2021). Consequently, this chapter's findings provided insights into the

practical limitations that hinder widespread implementation of output-based PWS.

5.0.1 Limitations

Across the three chapters, this thesis has identified limitations that reflect broader challenges in the literature. These limitations also offer avenues for future research to extend this work. The findings of the second chapter must be taken into consideration alongside one major caveat. Many studies where the estimated service could not be delineated using the CICES framework were discarded. The concerns of overlapping ES and hence potential double-counting thus remain a hindrance to secondary analysis of primary valuation estimates (Ojea et al. 2012). Further, although the CICES framework is designed to be flexible and hierarchical to account for various applications (Haines-Young and Potschin 2012), its use is not without challenges. This flexibility introduces a subjectivity when employed by practitioners that can affect the comparability of results across studies.

The findings of the third chapter are limited by missing records of key variables in the Survey of Drinking Water Plants (SDWP). For instance, the survey does not include information on the location of source water. Without this information, it is impossible to distinguish between upstream and downstream land near the source, which is expected to significantly affect the findings. Furthermore, if treatment facilities source water from distant locations, models using more spatialised scales may be misaligned in capturing the spatial dynamics involved. The sub-sub drainage basin boundaries delineated by the National Hydro Network (NHN) were large on average, making them less sensitive to such concerns. Finally, the models used in this study do not account for forest heterogeneity, or the potential cost savings provided by wetlands and grasslands. Including these factors could yield additional policy recommendations for sustainable land and water resources management.

The final chapter examined agri-environmental contracts using a standard setting in the literature. However, this treatment neglects many considerations that are relevant to PES programs. The model only considers a single-time period. Interactions in PES schemes are dynamic, with

contract often spanning over many years. Additionally, uncertainty regarding the flow of ES is also dynamic, as ES processes fluctuate in response to changing environmental conditions. With ongoing climate change, these variations will intensify, and so will the need for research that can account for this. For example, in real-world schemes, managing authorities can take steps to adjust their level of confidence in ES provision. Some PES schemes implement monitoring programs to establish baselines and improve knowledge of service provision (Corbera et al. 2007). Finally, the model's treatment of shirking and effort, though standard in the literature, is somewhat simplistic. The model could be extended to provide the landowner greater agency, such as by allowing them to choose their input level in response to price incentives set by the principal, as seen in the approach of Derissen and Quaas (2013).

The findings of the Millennium Ecosystem Assessment (MA) in 2005 have been adopted wholesale by researchers and policymakers. However, uncertainty remains a significant challenge in accurately valuing and managing ES across diverse contexts. We end briefly by considering the nature of uncertainty surrounding ES. Increasing our knowledge of ecological processes can improve the performance of PES schemes by enhancing contract design (Naeem et al. 2015). The first two chapters of this thesis demonstrated how rigorous methods can be employed to aid policymakers in understanding the benefits provided by forested watersheds. However, to believe that uncertainty surrounding the provision of ES can be entirely reduced is to fall for the trap of mistaking the conceptual metaphor provided by the ecosystem services concept as the only possible truth (Norgaard 2010). The final chapter offers insights into how uncertainty can be addressed using the tools currently available to economists. Together, the three chapters of this thesis explored practical ways to understand the benefits of forested watersheds, and where to go from here. One final note is to consider that many of the uncertainties in ES provision reflect deeper epistemological questions (Barnaud and Antona 2014). Managing uncertainty requires not only efforts to reduce it, but also recognising it can never be eliminated completely.

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